International Conference on Municipal Waste Combustion

April 11 – 14, 1989
The Diplomat Hotel
Hollywood, Florida
U.S.A.

CONFERENCE PROCEEDINGS

Volume 1

Congrès international sur l'incinération des déchets urbains

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COMPTES RENDUS

Volume 1

PROCEEDINGS

INTERNATIONAL CONFERENCE ON MUNICIPAL WASTE COMBUSTION

Hollywood, Florida USA
April 11-14, 1989

Volume 1

Sessions 1, 2A, 2B, 2C, 3A, 3B, 3C, 4A, 4B, 4C, 5A, 5B, 5C, 6

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ABSTRACT

The 1989 International Conference on Municipal Waste Combustion (MWC), sponsored by the United States Environmental Protection Agency and Environment Canada, is a special effort to provide an effective international forum for the transfer and exchange of technical information on municipal solid waste (MSW) incineration and ash treatment/disposal. Participating organizations include the American Society of Mechanical Engineers, the Air and Waste Management Association (formerly APCA), and the World Health Organization. The conference aims to provide an opportunity for professionals to meet and share ideas related to the planning, design, construction, operation, and maintenance of MWC facilities. This forum is also relevant to those involved in the regulation and research and development of new technologies addressing pollution problems related to MSW incineration.

The Conference Proceedings include papers examining relevant technical, regulatory, and social issues from the perspective of defining problems and proposing solutions, as well as describing experiences, forecasting trends, and reporting new and improved technologies. Papers and panel presentations are on the following topics: International Overviews; MWC Planning & Maintenance; Ash Characterization & Leaching; Risk & Health Effects; MWC Performance; Ash Treatment, Utilization, and Disposal; Sampling and Analysis; Residue Issues and Research Needs; Combustion Technology; Economic and Social Issues; Flue Gas Cleaning; Materials Recovery and Recycling; Quality Assurance/Quality Control, and Challenges of the '90's.

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PREFACE

The International Conference on MWC, held on April 11-14, 1989, in Hollywood, Florida, is sponsored jointly by the United States Environmental Protection Agency and Environment Canada. The Conference is devoted to the exchange of current research developments on MSW incineration and ash disposal. Education in MWC and ash disposal and the dissemination of technical information are the primary objectives of the meeting.

Presented papers and two panel discussions comprise the program. The papers and panel presentations address a wide range and variety of MWC issues: International Overviews; MWC Planning & Maintenance; Ash Characterization & Leaching; Risk & Health Effects; MWC Performance; Ash Treatment, Utilization, and Disposal; Sampling and Analysis; Residue Issues and Research Needs; Combustion Technology; Economic and Social Issues; Flue Gas Cleaning; Materials: Recovery and Recycling; Quality Assurance/Quality Control, and Challenges of the '90's.

The conference speakers include Environment Canada and United States Environmental Protection Agency personnel, as well as representatives from federal and state environmental/health agencies, research and development groups, academic personnel, and manufacturers. Participants from Belgium, Denmark, Federal Republic of Germany, Italy, Japan, Sweden, Switzerland, The Netherlands, and United Kingdom add an international perspective on MWC research and development plus related issues.

The co-chairmen of the conference are Theodore G. Brna of the United States Environmental Protection Agency's Air and Energy Engineering Research Laboratory and Raymond K. Klicius of Environment Canada's Industrial Programs Branch, Conservation & Protection.

The Conference Proceedings are presented in two volumes and include papers received by March 14, 1989. Also, included are abstracts of papers which will not be available until the conference. The contents of each volume are organized as follows:

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REFUSE INCINERATION IN EUROPE

by : A.G. Buekens
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ABSTRACT

Europe has played a major role in the development of current refuse incineration technology. Still, the technical standards and the relative importance of incineration in refuse disposal, as well as the attitudes towards this system are quite variable from one country to another.

The main steps determining the evolution of incinerator technology have been: the development of mechanical grates and of dedicated boiler technology and the advent of novel methods in flue gas cleaning and of automatic monitoring and combustion control systems.

Other developments worth of mention are: co-firing of refuse with sewage sludge or coal, preparation and firing of Refuse-Derived-Fuel, and pyrolysis and gasification. None of these options at present seems capable of challenging conventional mass burning.

The latter has been practised especially in densely populated regions, where landfill space is scarce. Nevertheless, public opposition against incineration has been vigorous and has prompted a lot of analytical and R & D-work, e.g. with respect to

- characterizing the content of refuse and of incinerator bottom-ash and fly-ash with respect to pollutants (heavy metals, topical organic contaminants),
- selective collection of hazardous and polluting components of refuse,
- a better understanding of the combustion process,
- a gradual improvement of flue gas cleaning technology.

As a conclusion, the main technical, environmental and managerial challenges of refuse incineration are highlighted and the future of refuse incineration is scanned.

INTRODUCTION

Municipal Solid Waste (MSW) collection and disposal is a worldwide problem, tackled according to local conditions and traditions, available means and desired service levels (1). Thoughout most E.E.C.-countries MSW is collected once or twice a week, sometimes even daily with compacting vehicles. The technical standards of disposal, the relative importance of incineration as well as the attitudes towards this option are quite variable from one country to another, however.

The least expensive MSW disposal method is landfill, with a wide range of disposal costs, from less than 5 $ for methodic refill of a pit to almost 40 $/tonne for systems featuring double barriers, biogas and leachate collection and treatment.

Composting has been extensively used in the Netherlands, in two large plants (Wijster, Mierlo), with MSW supplied by rail and in a few, now defunct, trommel or rasp type units. In France and Italy quite a few small composting & incineration plans have operated, with vineyards as a major compost market. Marketing of compost, stained with glass and plastic film and charged with heavy metals has been difficult, if not impossible in most E.E.C.-countries.

Incineration originated both in England and the U.S.A., in the second half of the 19th centrury. The first continental plant was erected in Hamburg early in the 20th century, after the City was exposed to a cholera epidemia impeding its refuse export to the countryside. Early Dutch and German plants were critisized because of grit and smoke emission; in those days the refuse was often barely auto-combustible.

During the twenties and the thirties numerous mechanical grate systems were patented for firing coal and also refuse. Refuse was fired on superimposed retractable grates (Heenan & Froude), reciprocating mechanical grates (Volund), or in shaft furnaces (Didier). Many plants featured a special element for clinker outburn, e.g. a rotary kiln (Volund), or a shaft resting upon a tiltable lock (Saint Vénien) (2).

After the second world war large cities were all confronted with a rapidly rising volume of refuse, a steady increase of its heating value and dwindling landfill space. During the fifties and sixties numerous cities hence resorted to refuse incineration, for large plants mostly with heat recovery, and in few cases in combination with coal or dried sewage sludge. Still it was discovered that refuse was more difficult a fuel than anticipated, and numerous plants suffered from severe boiler fouling and corrosion, which rendered a redesign of combustion chamber and boiler necessary. The use of electrostatic precipitators made the flue gas plume invisible and for a while refuse incineration was reasonably well accepted, even if it were the most expensive disposal option (2).

Still, environmental pressures were growing and analytical tools became more available, sensitive and refined. The first version of the German TA-Luft Code (1974) imposed the scrubbing of acid gases (HCl, HF) from the flue gas. The fly-dust, escaping from the stack, proved to be heavily enriched in some metals (Zn, Cd, Pb, ...) and in 1977 the presence of dioxins was demonstrated in fly-ash (3).

In Sweden refuse incineration was to be replaced by mechanical separation, yielding Refuse-Derived-Fuel (RDF) and compost, but soon it was realized that compost was unsaleable and refuse easier to burn than fluff RDF. In the F.R. of Germany the development of low temperature pyrolysis systems , e.g. Kiener and BKMI was spurred. After a while it was decided that the dioxin issue could be solved by (a) higher combustion temperatures and (b) deeper flue gas cleaning. In the meantime the mechanisms of dioxin generation were elucidated (cf. the presentations of prof. Hagenmaier, prof. Vogg and Environment Canada).

ATTITUDES TOWARDS REFUSE INCINERATION

MSW incineration is virtually absent in Greece, Portugal and Ireland. Mediterranean refuse is rich in organics and moist, which depresses the combustion temperature and leads to heavier organic emissions, including dioxins. Italy, because of the Seveso-accident, is quite sensitive to this problem; hence numerous old and smoky incinerator plants were closed.

The use of plastics was also attacked, although their presence tends to rise the combustion temperature (4). In the U.K. landfill of MSW continues to be hailed as a common-sense and cheap way of disposal, even when in co-disposal with specific hazardous wastes. Emission codes for the few existing MSW incinerators have remained relatively lenient.

In France the permitted emission levels are definitely higher than in Nordic countries, especially with respect to the numerous small incinerators having (at best) a simple grit arresting system. In Belgium there is a multitude of small plants, equipped, however, with E.S.-precipitators. In its Northern (Flemish) Region the importance of MSW incineration has augmented dramatically (from 28 % in 1981 to 51 % in 1987), merely by better regional planning and making a continuous usage of existing plant. Flue gas scrubbing has been imposed since 1983 and is being gradually implemented to all plants.

The F.R. of Germany, the Scandinavian countries and Switzerland have always adopted strict emission standards. As a consequence, numerous clean-up technologies were developed and have been applied in these countries (5).

INCINERATOR EMISSIONS

Incinerator emissions are due to (a) incomplete combustion and (b) the presence of pollutants in refuse.

Improved combustion control has allowed to decrease CO-emission levels by a factor of 10 or better (from 500-1000 to 30-100 mg/m^3, s.t.p.). Progress in this area has been achieved by (2) :

- a more careful mixing of refuse,
- mainly by providing a relatively high combustion temperature, a long residence time and adequate mixing,
- high speed injection of secondary air at the outlet of the primary combustion chamber, using series of narrowly spaced air nozzles. Also the first convection tube banks may help in this respect,
- aerodynamic study of the mixing processes in the combustion chamber and the radiative and convective boiler parts, using reduced-scale cold models or (more convincing) field measurements.

Even when complete, combustion of refuse is still a source of pollutants, such as HCl, HF, SO_2, NO and heavy metal containing aerosols. Pollution abatement, in principle, may be pursued by preventive actions, i.e. avoidance of polluting refuse compounds, or by flue gas cleaning.

Extensive analytical studies were performed in order to assess various pollutants in refuse and its fractions, e.g. in paper & board, plastics, vegetal matter, fines, or miscellaneous (i.e. mainly composite) compounds (Table 1 to 3). The available data clearly demonstrate, however, that none of these can be termed a "clean fuel". In other words, even wastepaper (and, a fortiori RDF), when fired, still needs extensive flue gas cleaning in order to attain present-day TA-Luft standards (Table 4). This was obviously a blow to many a RDF production or firing venture.

Special care was devoted to defining and quantifying the sources of dangerous heavy metals, e.g. Hg, Cd, Pb, (7). Measures were taken to replace cadmium or lead-based stabilizers and pigments. Also chlorinated refuse components (PVC, PCB-containing small transformers) have attracted some attention and many times it has been inferred, no doubt erroneously, that a ban on PVC-packaging would contribute seriously in solving the dioxin or the acid rain problem.

Selective collection of hazardous or polluting refuse compounds is not new, since it has been practised in Tokyo from 1973 onwards for compounds "unfit for incineration" (8). In a wider context, specific household wastes have been collected in Denmark, Sweden, F.R. of Germany, the Netherlands and Flanders. Table 3 lists a few such compounds.

It is still too early to assess the results and perspectives of such actions. In Sweden, for instance, the Hg-content of refuse decreased from 5 to 3 g/tonne by collecting button-type cells. Mercury is difficult to eliminate from flue gas and its collection efficiency is never complete (9). The selective collection and incineration of spent pharmaceuticals is more questionable, especially in areas were refuse is incinerated anyway. Moreover, collection, identification and disposal of special wastes is a factor 10-100 more expensive than standard.

Incinerator emissions are discharged to the atmosphere through huge stacks, typically of 60 m height or more. The resulting dispersion normally guarantees negligible supplemental immission levels. Still, in some cases the deposition of small charred particles, which are incompletely captured by E.S. precipitators, has been a problem. Moreover, poor housekeeping may be a source of fly-ash blowing around the plant.

GAS CLEANING

Before TA-Luft standards gas cleaning was limited to dust collection, normally by E.S. precipitators, in small plants by cyclone separators or (rarely) wet scrubbers.

Wet scrubbing of HCl allows to obtain low and even very low emission levels. Most early plants suffered severe operating problems, because of fast corrosion of any insufficiently protected tubes, wastewater treatment problems, mist carry-over, wet and corroding stacks, etc. Present scrubbers in either rubber lined steelwork or glass reinforced polyester are capable of reliable operation, but retrofitting and the related wastewater and sludge handling are often posing problems of limited space.

In the semi-dry processes several variants are possible:

- injection of a lime or soda slurry, with complete evaporation of the aqueous phase. This system works well as long as scaling problems can be mastered.
- using the neutralized wastewater, arising in a wet scrubber in (a) an evaporative spray cooling tower, preconditioning the flue gas or (b) an off-line multiple effect evaporator. The latter was shown capable of producing reusable grades of kitchen salt.

A much simpler solution still is the dry process, based on the pneumatic injection of free flowing, neutralizing powders, normally $Ca(OH)_2$ or CaO, more seldom $NaHCO_3$.

Some advantages and disadvantages of the systems are summarized in Table 5.

Low emissions can only be attained during trouble-free plant operation. Some problem areas and remedial actions are summarized in Table 6 (2).

RESIDUE DISPOSAL

Incineration literally reduces MSW to cinders thus lowering its volume, putrescibility, health, flammability and toxicity hazards. An E.E.C.-City with 1,000,000 inhabitants typically generates

1,000 tonnes of MSW a day
or 6,000 m^3 of MSW a day (standard density)
or 2,500 m^3 of MSW a day (compressed)

Whereas the combustion residue amounts to

250-400 tonnes a day (a reduction of 60-75 wt. %)

or 200-320 m^3 a day (a reduction of 95-97 %, based on original volume)

In the lower countries of Europe, e.g. the Netherlands and the North of the F.R. of Germany, there is a tradition of eliminating ferrous metal, grading the bottom ash into various size ranges and crushing large sintered aggregates. Markets are to be found as a low grade substitute for sand or gravel, used int. al. for hardening paths or as a sublayer for secondary roads.

The fly-ash, on the other hand, has never found any kind of usage, because of its acid properties and variable composition.

Twenty years ago Lurgi built a fly-ash sintering plant in Berlin, which operated for a very brief period only. Various cold methods have since been proposed for solidifying or immobilizing the heavy metals contained in fly-ash, and for leaching out the heavy metals content with spent acid.

A number of leaching tests have been proposed with the aim of assessing the potential hazards of fly-ash. Leaching may take place with distilled or acidified water, or with synthetic landfill leachate. Testing is either by stirring with eluant or by using packed column contactors (usually for mixtures with bottom-ash). Many institutes have conducted research in this field, without succeeding, however, of establishing any convincing relationship between laboratory testing and field practice.

Obviously, the common sense approach is to consider bottom-ash as a relatively inert fill material, that still requires some monitoring after being dumped in a suitably designed landfill site. More precautions are required when filling fly-ash, to prevent its dispersion by the wind and reduce leaching. Boiler ash consists of coarse particles with intermediate composition. Hydroseeding has proved to be a proper method for developing growth on (coal) fly-ash landfills.

OUTLOOK

Refuse incineration is a well established technique to reduce the volume of MSW, leaving a sterilized, non-putrescible bottom-ash, a fly-ash markedly enriched in heavy metals and, as flue gas codes become more stringent, supplemental liquid or solid waste streams arising in flue gas treatment.

Incineration requires heavy initial investment, relatively high operating and maintenance costs and skilled managers and operators. Its relative position in MSW disposal may be expected to increase, because of still higher environmental and social pressures on landfill. Incineration is regarded as a method of resource recovery; a major problem with respect to heat recovery and utilization is institutional, i.e. the monopolistic structure of power companies in many E.E.C.-countries.

On the basis of objective cost data, one would expect a wide application of sanitary landfill in the countryside, local hydrogeological conditions permitting, and of incineration in densely populated, industrialized regions. The potential for incineration, moreover, would logically be lower in the Mediterranean countries, with a moist and much denser refuse.

It seems unlikely that actual emission standards will continue upon the present trend of tightening. It should be realized that when codes are really tight, emissions are more likely to occur from plant upsets or poor housekeeping than from baseline operation. Hence, it is to be expected that

- in the more advanced E.E.C. countries the monitoring effort as well as the pressure to avoid incidents will increase and generalize,
- the countries with more lenient codes will be under strong moral pressure to move along similar lines as the other, albeit with an effective time lag of the order of 10 years.

The supplement cost for cleaning flue gas is likely to be of the order of 8-10 \$/tonne of MSW, a supplement many citizens are prepared to pay for a cleaner environment. It is also argued that rendering waste disposal more expensive contributes to reduce generation rates, but such an influence at present seems neither plausible nor perceptable.

Two and three field electrostatic precipitators are preferred as long as moderate dust levels can be tolerated (100 or 50 mg/m3). Lower levels require supplemental volume and investment. Baghouses, on the other hand, are preferred in dry and semi-dry systems. The layer of limepowder retained assists notably in lowering acid gas and heavy metal emission levels.

Wet scrubbers are capable of achieving almost any acid gas emission code, provided they feature efficient double or multiple stage demisters. Removal of HCl proceeds well at low pH-values, but removal of SO_2 requires a neutral wash with a pH = 6 to 8; NO_x- removal is still at an experimental or demonstration stage. In general, the limiting heavy metal concentrations are respected without problems. For mercury removal, however, a deep cooling (i.e. wet scrubbing) by means of an acidic liquor is desirable.

The residual waste streams arising in MSW incineration and consecutive flue gas treatment are quite different with respect to the environmental hazards and challenges they provoke. Bottom-ash is generally considered to be unproblematic. Fly-ash often requires special care, even solidification to reduce leaching of heavy metals. The spent lime and the neutralized salty scrubbing liquors still pose a problem.

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TABLE 1. AVERAGE COMPOSITION OF HOUSEHOLD REFUSE IN THE FEDERAL REPUBLIC OF GERMANY (6)

	Weight Fraction
Paper, printed matter	14.6
Board	4.1
TOTAL A (PAPER FRACTION)	18.7
Paper composites	1.2
Plastics	6.1
Textiles	1.5
Wood, leather, horn, bones, rubber	2.3
TOTAL B (PLASTICS FRACTION)	11.1
Fines (siftings ≤ 8 mm)	8.6
TOTAL C (FINES)	8.6
Vegetal matter in sieve in fraction above 40 mm	26.8
Siftings 8 to 40 mm	15.6
TOTAL D (VEGETAL MATTER)	42.4
Ferrous metal	3.5
Non-ferous metal	0.4
Glass	11.6
Minerals	2.9
Composite objects, e.g.	0.8
TOTAL E (NON ANALYSED)	19.2

Data obtained in 1980

TABLE 2. TRACE CONTAMINANTS IN VARIOUS REFUSE FRACTIONS (6) , ppm

Fraction	A	B	C	D	A to D
Element	Paper Fraction	Plastics Fraction	Fines	Vegetables	Analysis Refuse
S	900	1575	110	875	1050
Cl	2925	21525	1325	1375	5025
F	31	14	30	3.5	17.5
Cd	1.5	14.5	2.7	4.3	5.0
Cu	100	525	26.5	575	350
Fe	2000	3875	12050	7025	5950
Hg	0.15	0.42	0.32	2.0	0.91
Mn	75	100	625	300	275
Na	1550	1475	2500	3575	2500
Pb	125	800	550	900	
Zn	375	975	1125	750	

Sources of Various Elements

Cl	Plastics, wood, rubber, leather
Al	Fines, glass, mineral fraction, paper
Ba	Paper, board, glass, mineral fraction
Cd	Plastics
Cr	Wood, rubber, leather
Pb	Plastics, fines
Zn	Wood, rubber, leather, paper &board

TABLE 3. SPECIAL COMPOUNDS IN HOUSEHOLD REFUSE

	Metals					Basic Lyes	Acids	Solvents	Organo-halogens	Tar derivatives
	Cd	Hg	Ni	Pb	Zn/Cr					
Batteries	x	x	x	x			x			
Biocides								x	x	
Cleaners						x	x	x		
Cosmetics								other chemicals		
Detergents								other chemicals		
Fluorescent Tubes		x								
Medicines								other chemicals		
Paints, Lacquer	x			x	x			x x		x
Paints removers						x		x	x	
Rust preventers				x	x			x	x	
Thermometers		x								
Wood preservation								x	x	x

Adapted from : Schenkel, W. Kassing, D. Entsorgung 2000 Leitfaden für Kommunen, Wirtschaft und Politik, Borner Energie-Report Verlags GmbH Bonn 1988

TABLE 4. LIMIT VALUES IN THE AUTHORITATIVE GERMAN TA-LUFT EMISSION STANDARDS

	Version 1974/83 11% O_2	At present	Units
SO_2	-	100	mg/m3 (s.t.p.)
Cl^- (HCl)	100	50	mg/m3 (s.t.p.)
F^- (as HF)	5	2	mg/m3 (s.t.p.)
dust	100	30	mg/m3 (s.t.p.)
class I	20 [a]	0.2 [d]	mg/m3 (s.t.p.)
class II	50 [b]	1.0 [e]	mg/m3 (s.t.p.)
class III	75 [c]	5.0 [f]	mg/m3 (s.t.p.)
CO	1000	100	mg/m3 (s.t.p.)
organic C	-	20	mg/m3 (s.t.p.)
dibenzodioxins		minimal	
dibenzofurans		minimal	
PCB, PCH, PFH, PBrH		minimal	

[a] Hg, Cd, Pb, Cr (VI), F, Ni, Se, Te, Th, U,V and their compounds (*) for ≥ 0.1 kg/h

[b] Sb, Ba, Co, Ag, Sr,Zn,..., for ≥ 1 kg/h

[c] B, Cu, Mo, Bi, W,... for ≥ 3 kg/h

[d] Cd, Hg, for ≥ 1 g/h

[e] As, Co, Ni, Se, Te, for ≥ 5 g/h

[f] CN^-, Cr, Cu, Pb, Sb, Sn, for ≥ 25 g/h

(*) in some only the water soluble compounds are concerned

TABLE 5. FLUE GAS CLEANING METHODS

Scrubbing method	Wet	Semi-dry	Dry
Operating temperature	45-70 °C	200-300°C	150-300°C
	low	high	medium [a]
	residual vapor pressure of mercury and cadmium salts and organic micropollutants		
Removal Efficiency			
for HCl, HF,	very high	fairly high	medium
SO_2	high, alkaline wash	low	negligible [b]
Residual Streams	- wastewater to be neutralized & settled - neutralized brine to be cleaned form heavy metals - sludge, to be thickened & dewatered	- dry residue consisting of spent lime & fly-ash	large exess of unreacted + spent lime
Operating Problems	- corrosion	- scaling	- solidification of residue

[a] medium, because of a large adsorptive power of the reactants

[b] unless lime is injected at 850°C

TABLE 6. MAIN INCINERATOR PROBLEM AREAS (2)

Area	Problem	Remedial action
Bunker	fires	- excavation of smouldering mass followed by soaking
	dust	- water atomizing - pressurized mercury lights
	inadequate capacity	- even out supply of MSW - increase furnace capacity
Shaft	obstruction	- shredding or shearing of bulky waste
	fire	- refractory lining, or water jacket - draft control
Furnace & Grate	refuse drying, ignition, burn-out	- air preheating, flue gas recycling - adequate residence time on the grate
	regular feeding	- feeding table or ram, with stroke period & length variable
	uneven refuse bed	- grate action, conveying, tumbling
	slagging of furnace walls	- air or water cooledwalls
	blocking of grate bars	- better grate design
	false air	- dust collection - closed construction - close dry extractor - avoid uncovered grate sections
Extraction of clinker ash		
- wet	wear, corrosion	- materials selection
- dry	wear, dust	- draft control
Radiative flues	corrosion	- avoid alternating oxidizing and reducing conditions - clad with refractory lining;
	fouling	study flow lines & mixing
Convective flues superheater	corrosion	- reduce temperature of flue gas and of steam - improve burnout of flue gas
	fouling	- provide one or more turns ahead of superheater or a - small evaporator tubebank
Economizer	corrosion	- increase preheat temperature - avoid intermittent operation

Electrostatic precipitator	corrosion	- avoid condensing hygroscopic salts - avoid cold spots
	poor results	- avoid overloading - eliminate coarse particles with cyclones - avoid inleaking of air through hoppers
	fly ash solidification	- avoid glowing particles or hygroscopic effects
Wet scrubber	corrosion	- protective coating imperfect
	fouling	- study air/water flow distribution
Stack	corrosion	- avoid cold stacks - specify acid proof interior lining
Fans	equilibrium loss	- rebalancing

THE PROPOSED EEC DIRECTIVES ON THE CONTROL OF EMISSIONS FROM MUNICIPAL REFUSE INCINERATION AND THEIR IMPLICATIONS FOR MEMBER STATES

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ABSTRACT

In March 1988 the Commission of the European Communities proposed two Directives on the prevention of air pollution from municipal solid waste incineration plant. These proposals sought to introduce strict controls on emissions of a wide variety of pollutants, both for new and existing plant.

Ensuing discussions have led to a very fruitful debate on the different approaches to incineration control which exist within the EEC and have profoundly influenced the forward planning of many member states. This has been particularly noticeable on the control of combustion conditions for the minimisation of dioxin emissions, the specification of emission limits, the means of achieving them, monitoring philosophies and the ways of implementing the decisions.

This paper compares some of the contrasting approaches which have been considered and goes on to examine the implications for the member states. It also attempts to indicate how and where the debate has influenced R&D.

BACKGROUND

In March 1988, after nearly two years of preparatory work, the Commission of the European Communities proposed two Directives on the prevention of air pollution from municipal solid waste (MSW) incineration plant (1). These proposals sought to introduce controls on combustion conditions, emission limits for a wide range of pollutants and specified monitoring provisions. The object was to ensure good operational practice and limit emissions of particulate material, acid gases, heavy metals and dioxins. In fact it was largely as a result of concern over dioxins that a process with mainly local effects became the subject of a directive at all. A Best Available Technology Not Entailing Excessive Cost (BATNEEC) approach was proposed for all new incinerators and a timetable was put forward for a stepwise upgrading existing plant to those standards. These proposals contained a number of features new to the Community and the negotiations have been extensive. Work on the New Plants Directive is nearing completion and will be followed by work on the Existing Plant Document.

The original proposals for new plant are shown in Table 1 and those for existing plant in Table 2. Debate is still continuing and details of the proposals have changed somewhat but the major themes remain and this paper will attempt to elaborate on the main points and show where the discussion has implications for member states. It will also try to indicate the R&D which has resulted or is likely to be required.

COMBUSTION CONDITIONS

When the possibility of Community action was first mooted the concern over dioxins in member states was such that the Commission felt bound to act and since monitoring was at an early stage of development a preventative approach was thought to be the most fruitful way to proceed (this was somewhat contrary to the usual form of directives which specify the desired result of action rather than how it should be achieved). At that time combustion conditions were thought to be of primary importance, indeed the Italians, as a result of their own work in the area, were well into their programme of installing secondary combustion chambers. So after consultation with member states a condition of at least 2 seconds at 850 centigrade was put forward, largely influenced by the Danes. Auxilliary burners were also specified to ensure that minimum temperatures were maintained at all times. This is considerably different from the recommendations of at least 1 second at 1000 degrees currently being put forward in the

USA and Canada (2) but has the advantage that NOx emissions should be somewhat lower. There remains the question as to how the temperature and residence conditions will be defined.

In the course of preparing and negotiating the new plant directive our understanding of the formation of dioxins has progressed, largely as a result of the ongoing work in the Federal Republic of Germany, and it questioned whether it was necessary to specify combustion conditions at all if care was taken over particulate arrestment and rapid cooling of the flue gases. On the other hand there is some evidence that the amount of dioxin formed is proportional to the amount of carbon in ash, so a counter theme has been that the relationship between composition of the material leaving the combustion chamber and that found post abatement is still not fully understood. Also dioxins are only one of a number of possible products of incomplete combustion and controls on combustion would serve to minimise emissions of a variety of potentially toxic carbon based components. Consequently it is likely that the combustion condition requirement will be retained.

Since specific combustion controls would only be relevant to conventional technology a separate article was introduced to ensure that innovation, both in technology or in how conventional plant were operated, would not be stifled, provided that the resulting emissions could be demonstrated to be equivalent to those achieved using conventional technology operated in accordance with the Directive.

The debate has led to several combustion related programmes aimed at providing information for policy makers. The Danes have an on-going programme looking into the mechanism of dioxin formation and the UK is looking into the technical problems of measuring and defining combustion reference conditions to name but two.

POSTCOMBUSTION ABATEMENT

As can be seen from Table 1 emission limits were originally specified for total dust, metals and acid gases. Levels were set which were thought to be representative of BATNEEC, in this case the equivalent of a three field electrostatic precipitator and some form of acid gas scrubbing. It remains to be seen whether the intermediate requirements for existing plants are acceptable to Member States (Table 2) and it is interesting that a more pragmatic approach is apparently being taken to updating plant in the USA.

The debate has been vigorous on the issue of particulate control, it has been questioned whether 50 mg/Nm3 is adequate considering the importance of dioxin and heavy metal emissions, 30 mg/Nm3 is thought more suitable by a number of the member states. It is unlikely to be cost effective to use electrostatic precipitators to achieve this; similarly if dry scrubbing systems are used there might also be difficulties. Consequently the debate has, to some extent, reduced to whether the 20 mg improvement is sufficient to justify the reduction in the choice of abatement equipment now facing the operators - the point is yet to be resolved.

How to deal with small plants is proving a problem, the original proposal was for relaxed limits for small plant in isolated areas subject to high seasonal variability in load. The particulate loadings originally specified envisaged very high efficiency cyclones or small EP's but it has been argued very strongly that the community should not encourage the proliferation of small plants because they are more difficult to control and do not justify a high level of expenditure on control.

EMISSIONS MONITORING

The monitoring provisions have been the subject of more scrutiny than possibly any other provision. Underlying the discussion has been the questions of what is the purpose of the measurements are they for control purposes or for checking compliance retrospectively and what should be the averaging period? On several of these issues decisions are yet to be reached. Temperature, oxygen/carbon dioxide, carbon monoxide, particulates and HCl etc. are, or could be, control parameters and should be determined frequently and acted upon rapidly. On the other hand the emissions of metals etc will be a function of the efficiency of the dust arrestment equipment and need not be determined so frequently. These issues can be expected to lead on to discussions of when a breach constitutes a breakdown which should lead to plant shutdown.

Whether or not to monitor dioxin emissions has also been a contentious point. Which species are determined and by which method is a difficult question and it is expected that the Commission will come up with proposals by 1990 consequently this is a likely area of continuing R&D within the Community.

Standards for the measurements proposed and a mutually acceptable system of testing and accreditation for instrumentation which will allow instruments made and approved in one member states to be used in another is another problem still to be resolved. Possibly testing along the lines of that carried out by TÜV in the FRG might be the answer. Some work is already being done on intercomparability of dioxin analysis in member states under the auspicies of the Bureau of Community Reference. There is considerable scope for cooperative action with other international groups in the general area of method standardisation.

IMPLICATIONS FOR MEMBER STATES

An approximation of the number and size distribution of MSW plant in the EEC is shown in Figure 1. It can be seen that the size and type of plant used varies considerably illustrating the wide variety of national/local needs which make generalisations from country to country difficult. France has a large number of plant, ranging from the very small to the very large; the population distribution is such that a lot of small plant are needed to serve scattered populations in areas where landfill is difficult but very large incinerators are needed to serve the densely populated cities. Germany, the Netherlands, and the UK have relied mainly on large plant with Belgium, Denmark and Italy coming somewhere in between. Ireland, Portugal and Greece have few if any plants and Spain is gaining an increasing number.

It can be seen from this breakdown that the Directive proposals have very different implications for different member states. France, for example, faced with a large number of small plants faces the difficulty of equating the cost of provisions largely aimed at large plant with the gains which can be achieved from small plants. Consequently the French have been examining pollution control systems especially suitable for smaller plant particularly in the field of gas scrubbing.

The Italians, as a result of Seveso, are particulary sensitive over the dioxin issue and are keen to ensure that every precaution is seen to be taken to prevent dioxin emissions. Consequently at home they are reducing the number of small plants and have tightened up on combustion conditions. They are keen to see any EC proposals strong in these areas. They would also like to see routine dioxin monitoring carried out, an area where they have a great deal of R&D experience.

A number of states have expressed doubts as to the wisdom of allowing a proliferation of small plants but for a number of the Community there is little or no alternative as Spain has been at pains to point out. Work on the emission characteristics of small incinerators is seen as of particular value and the UK for one has an ongoing programme in this area. In view of the ramifications for hospital and special wastes I think this is another area where international cooperation would be of value.

PROBLEMS OF IMPLEMENTATION

Many of the proposals will require considerable readjustments of national legislation and I will comment on the UK case as the one which I am most familiar with. Even though the UK was reviewing its air pollution control legislation the EC Proposals nevertheless called for rethinking in a number of areas. It now seems likely that up to half the thirty eight MSW incinerators will be closed down because they are too old and inefficient to warrant refurbishment to the standards required. A further ten or so will only be refitted sufficiently to allow a five year extension to their life while further decisions are made. Only a handful of the largest and most modern plants are certain to continue and these are at an early stage of planning and costing major retrofit programmes. The implications for UK waste disposal policy are also considerable especially with pressure coming to bear on landfill so much of the UK R&D programme is geared around programmes to improve the performance of existing plant, improving instrumentation and control, and encouraging better operational practice.

If anything the difficulties of adjusting national legislation to allow a harmonised approach across Europe has been the most challenging problem of all. But the will is there and it is to be hoped that these moves increase public confidence; for it is only then that incineration will be able to take a proper place in our waste disposal and energy policies.

REFERENCES

1) Commission of the European Communities, Proposal for a Council Directive on the Prevention of Air Pollution from New Municipal Waste Incineration Plant, Proposal for a Council Directive on the Reduction of Air Pollution from Existing Municipal Waste Incineration Plant. COM(88)71 Final, Brussels, March 1988.

2) Le Conseil Canadian Des Ministres de Resources et de l'Environnement. Operating and Emissions Guidelines for Municipal Solid Waste Incinerators, Oct 1988.

TABLE 1 NEW PLANT DIRECTIVE

- Total dust	
. nominal capacity equal to or greater than 5 tonnes per hour	50 mg/Nm^3
. nominal capacity less than 5 tonnes per hour	100 mg/Nm^3
- Heavy metals	
. Pb + Cr + Cu + Mn	5 mg/Nm^3
. Ni + As	1 mg/Nm^3
. Cadmium (Cd)	0.1 mg/Nm^3
. Mercury (Hg)	0.1 mg/Nm^3
- Hydrochloric acid (HCl)	
. nominal capacity equal to or greater than 5 tonnes per hour	50 mg/Nm^3
. nominal capacity less than 5 tonnes per hour	100 mg/Nm^3
- Hydrofluoric acid (HF)	
. nominal capacity equal to or greater than 5 tonnes per hour	2 mg/Nm^3
. nominal capacity less than 5 tonnes per hour	4 mg/Nm^3
- Sulphur dioxide (SO_2)	300 mg/Nm^3

emission limit values, standardized at the following conditions : temperature 273 K, pressure 101.3 kPa, 11% oxygen or 9% CO_2 and dry gas, shall apply to new municipal waste incineration plants :

TABLE 2 EXISTING PLANT DIRECTIVE

a. Plants whose nominal capacity is at least 6 tonnes of waste per hour	
- total dust	100 mg/Nm^3
b. Plants whose nominal capacity is less than 6 tonnes of waste per hour but at least one tonne per hour	
- total dust	150 mg/Nm^3
c. Plants whose nominal capacity is less than one tonne of waste per hour	
- total dust	600 mg/Nm^3

emission limit values, standardized at the following conditions : temperature 273 K, pressure 101.3 kPa, 11% oxygen or 9% CO_2 and dry gas, shall apply to existing municipal waste incineration plants :

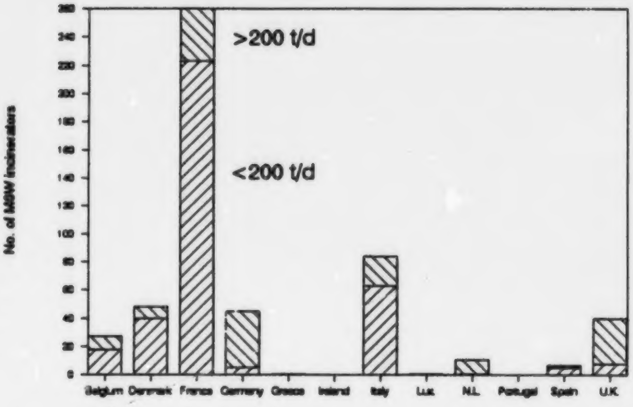

Figure 1. Incineration in the EC

WHO VIEWPOINT
ON HEALTH IMPACT FROM MUNICIPAL WASTE COMBUSTION

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ABSTRACT

The WHO Regional Office for Europe has examined and evaluated the health hazards of emissions from the combustion of sewage sludge and municipal solid waste. This work concerned itself with dioxins, furans, polynuclear aromatic hydrocarbons (PAHs) and heavy metal compounds. It was concluded that, in the case of modern well maintained and well operated incinerators, these chemicals will not be of major concern to human health. However, because the heavy metals will concentrate on the fine ashes to be disposed on landfills, they will require some additional attention to prevent their seepage through the soil into the waste aquifer.

HEALTH FOR ALL

About ten years ago, in September 1978, the Member States of WHO, worldwide, came forward with a declaration to achieve "Health for All by the Year 2000" (the Alma-Ata Declaration). Following this Declaration, a global strategy was formulated by WHO, including the development of indicators for monitoring progress towards this goal. This has included guiding principles for a managerial process for national developments and a health programme evaluation. To assist the European Member States to achieve this goal, the WHO Regional Office for Europe (WHO/EURO) came up in 1984 with a plan of action for implementing the global strategy in the European arena. This included the formulation of 38 targets in support of the European regional strategy for health for all by the year 2000 (1). Target 23 has been dedicated to the control of hazardous waste, while municipal solid waste (MSW) and municipal sewage sludge (MSS) have not anymore been mentioned. The reason has been, that WHO/EURO was dealing with MSW problems for a number of decades and thought that perhaps it could not anymore make a significant contribution in this area. At the same time, hazardous waste issues have become increasingly significant, and have become a popular subject in the news media and among politicians due to a number of now very wellknown disasters (such as Love Canal).

WHO ACTIVITIES ON MUNICIPAL WASTE

However, despite the present emphasis on hazardous waste, WHO/EURO has found itself actively involved in recent years in issues concerning MSW and MSS. In the past, several expert meetings were undertaken with reference to MSW and MSS. The products of some of these past activities were gathered into one book, which includes a comprehensive chapter on MSW incineration (2). Another activity dealt with the management of waste from hospitals and other health care establishments, including combustion (3).

The recently growing emphasis worldwide on combustion of MSS and MSW has led to certain requests from Italian regional authorities. Consequently, two meetings were organized by WHO/EURO in 1986 and 1987 on emissions from such combustion with particular regard to dioxins, furans, polynuclear aromatic hydrocarbons (PAHs) and heavy metal (HM) compounds (4, 5, 6, 7). A specialized seminar on organic emissions from municipal incinerators was held in January 1987 in cooperation with the International Solid Wastes and Public Cleansing Association (ISWA) and its Danish member society DAKOFA (8).

SUMMARY OF THE SITUATION

MSW collection and the handling of MSS are now common practice throughout most of the European Region, particularly in urban areas. Yet, although great efforts have been made to improve methods for handling MSW, such as segregation of recoverable materials (like metals, paper and glass), and its treatment thereafter, difficulties in the final disposal of waste have increased during recent years. Whereas in the past MSS was often dumped into the sea, when opportune, this practice is generally not anymore acceptable and permitted. Landfilling, being one alternative for both MSS and MSW , run also into troubles, sometimes because of pollution of groundwater, but more often because of the growing shortage of suitable sites near populated areas. Also, public dissatisfaction at perceived eyesores, with problems due to rodents, birds, and sometimes human scavenging has limited the options for landfilling sites. Waste combustion has therefore often become a major disposal alternative, although only now well operated and maintained incinerators have shown to be satisfactory with respect to minimizing hazardous emissions to the atmosphere.

However, emissions from MSW and MSS combustion have long aroused concern. In the last decade, polychlorinated dibenzo-para-dioxins (PCDD) and dibenzofurans (PCDF) were reported in fly ash from MSW incinerators. Similarly, HM compounds and organic compounds, such as PAHs, have more recently also given cause for concern. Consequently, the general public, having become aware of the high toxicity of these compounds, has given much attention to combustion of MSW and MSS. Obviously, there has been a need to develop a better understanding of the emission levels and potential public health effects of these ingredients of air pollution from municipal incinerators.

PATHWAYS, EXPOSURE AND HEALTH EFFECTS

Dioxins and Furans (for detailed literature list see (4))

PCDD and PCDF emitted from MSW and MSS combustion are often found in the environment adsorbed onto particulate matter, and may also contaminate water bodies and plants. Photodegradation is likely to be one factor affecting the ultimate fate of PCDD and PCDF. In view of their lipophilic nature, persistent PCDD and PCDF have a potential to accumulate in the food chain, being their major human exposure pathway.

To assess the impact of PCDD and PCDF emissions from MSW and MSS combustion on a local general population, one would have to examine the potential contribution of such emissions to the overall background daily dose. Emission exposure can be direct, through inhalation, dermal

INCINERATION OF MUNICIPAL SOLID WASTE REGULATORY INITIATIVES IN CANADA

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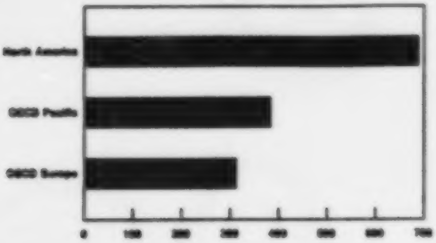
On a global basis the amount of waste associated with established lifestyles remains on the upswing. In a recent report the Organization for Economic Cooperation and Development (OECD) compared 1980 waste generation rates for member countries. These rates were broken down into three major (Figure I) geographical areas. While data base inconsistencies may be a factor in such a comparison it is noteworthy that Canada and the United States are the most prolific waste producers and have perhaps the most similar lifestyles.

Figure I - Amounts of Municipal Waste
(kg per capita, 1980)



In examining waste disposal practices (Figure II) it is evident that other factors influence trends. The OECD Pacific data (heavily influenced by

Figure II - Municipal Wastes Disposal Practices

	Compost	Incinerator	Landfill
N. America	—	4%	96%
OECD Pacific	1%	47%	52%
OECD Europe	6%	30%	64%

Japan) and OECD Europe both show the significant role incineration plays in waste management as well as a lower dependency on landfill as compared to North America. While the incineration trend in Europe has slowed somewhat, the North American projection shows very significant growth. It is these projections, and the recognition that emissions from incinerators may pose significant health and environmental impact, that has caused the development of major incineration evaluation programs.

In 1984, the Canadian Ministers of Health and Welfare and Environment issued a report from their Expert Advisory Committee on Dioxins identifying municipal solid waste incinerators as potential major sources of polychlorinated dibenzo-p-dioxins (PCDD) and polychlorinated dibenzofurans (PCDF) and asking for the development of measures to investigate this finding and, if necessary, the establishment of appropriate control measures.

In response to this directive Environment Canada developed the National Incineration Testing and Evaluation Program - NITEP. The objectives of NITEP have been stated as:

- ° to ensure parameters of concern with respect to environmental and health impact are identified and monitored for MSW combustion systems;
- ° to establish verified sampling and analytical protocols and to identify and evaluate the potential for use of surrogate parameters;
- ° to fully characterize combustion system design and operation in relation to emissions and energy output;
- ° to ensure all combustion and emission control processes likely to be proposed for use are examined;
- ° to establish suitable regulatory recommendations and controls where appropriate.

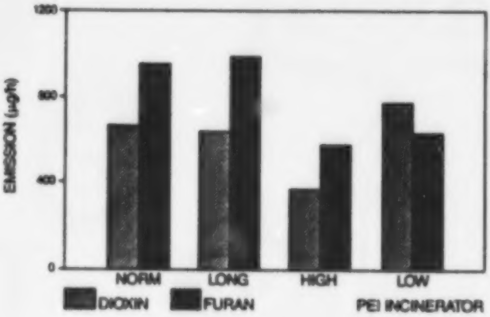
In 1984 NITEP was begun. In addition to laboratory activities and a supporting ash residue management research program, NITEP established four major field projects to develop the data base necessary for regulatory control proposals.

TWO STAGE COMBUSTION

The first technology to be considered was the smaller two stage combustion process often found in smaller communities - in this case Parkdale, Prince Edward Island. This facility is designed to burn 100 tonnes per day of MSW and produces steam for two adjacent hospitals. There is no emission control system and ash residues are disposed of in a municipal landfill. The details of this work are reported in EPS 3/UP/1 and the reader is referred to this document and supporting volumes for detailed test information.

One of the notable results (Figure III) was that for the four conditions tested (in triplicate) - representing the full practical operating window of this technology - there was no more then a factor of two between the best and worst case of PCDD emission (75 ng/Nm^3 vs 150 ng/Nm^3: - 100 ng/Nm^3 avg). For this technology it seems that the basic design is more of a factor than the specific operating condition.

Figure III - Total Dioxin/Furan Emissions by Group Mean



Of significance during these tests was the early observation that there was relatively little PCDD/PCDF exiting the high temperature zone of the furnace and that these levels increased by an order of magnitude across the boiler (Figure IV). Subsequent tests by others has attempted to explain this formation phenomena.

Figure IV - Concentrations of Organic Compounds at Boiler Inlet in Comparison to Stack

	Concentration (ng/Nm^3 @ 12% CO_2)					
	Long Cycle		High Temperature		Low Temperature	
Compound	Boiler Inlet	Stack	Boiler Inlet	Stack	Boiler Inlet	Stack
PCDD	0	107	1	62	14	123
PCDF	0	136	0	93	10	98
PCB	11*	38	8*	0	61	126**
PAH	10 475	8 010	12 226	6 633	29 481	12 490**
Chlorophenol	486	3 773	479	2 706	2 941	6 391**
Chlorobenzene	6 395	3 161	4 893	3 968	5 906	4 884**

* only one non-zero value (two readings were below detection limit)
** based on average of 2 tests only.

Overall, this technology emitted relatively low levels of particulate, PCDD/PCDF and other organics and metals for an uncontrolled unit.

AIR POLLUTION CONTROL TECHNOLOGY

In cooperation with Flakt Canada and the Quebec Urban Community, Environment Canada built a large scale pilot plant to study the capabilities of an MSW incinerator emission control system based on lime scrubbing and a fabric filter. The system built for this purpose (Figure V) gave operating flexibility to study the various combinations of process units. A full series of reports is summarized in EPS 3/UP/2 and the reader is referred to this literature for detailed information.

Figure V - Flakt's Dry System

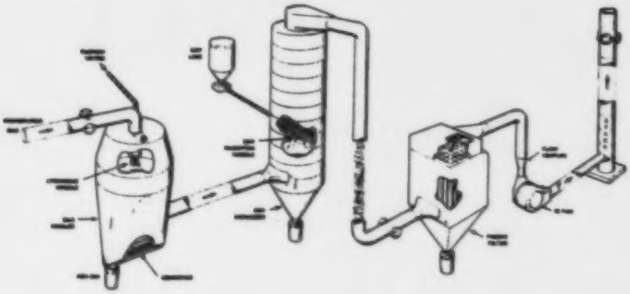


Figure VI - PCDD Concentrations in Flue Gas

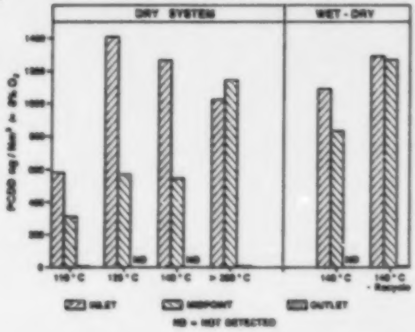


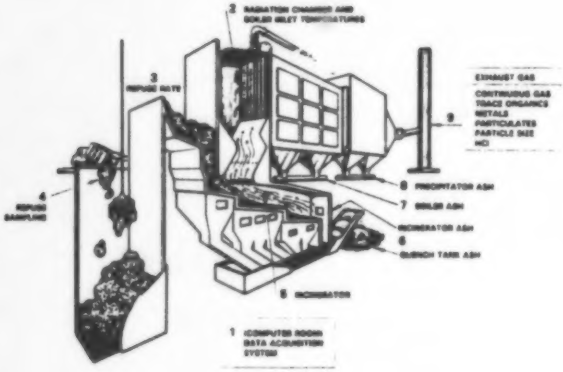
Figure VI gives an indication of system performance for six operating conditions (duplicated) tested during this program for PCDD. The two basic systems tested - one with a flue gas conditioning system (temperature control through humidification) followed by dry lime addition (noted as the "dry" system) and the second, wherein both lime and temperature needs are fulfilled through a single addition of the appropriate lime slurry, did not display sufficiently different performance to promote one over the other. Both were capable of virtual elimination of PCDD/PCDF in flue gas emissions but a more detailed assessment of the two processes would be required to establish process precedence. Overall metal removal was very high (99.9%+) with no significant difference between operating conditions. This is most likely attributable to the fact that exhaust gas particulate concentrations consistently were extremely low (< 5 mg/Nm^3).

The major metal observation was on mercury where temperature was the most significant parameter controlling final emission concentrations.

MASS BURN

For the NITEP mass burn tests, the Quebec Urban Community (CUQ) retrofitted to state-of-the-art status one of their four mass burn incinerators. This included full automatic process control. The old 2 field ESP was retained and left unchanged. Further changes to this equipment would involve scrubbers and fabric filters and were not a practical modification within the testing schedule. Figure VII shows the basic process as tested (Full reporting of these tests are in EPS 3/UP/5 and supporting volumes.)

Figure VII - Quebec Incinerator Sampling Locations


2 RADIATION CHAMBER AND BOILER INLET TEMPERATURES
3 REFUSE RATE
EXHAUST GAS
CONTINUOUS GAS
TRACE ORGANICS
METALS
PARTICULATES
PARTICLE SIZE
HCl
9
4 REFUSE SAMPLING
8 PRECIPITATOR ASH
7 BOILER ASH
INCINERATOR ASH
6
QUENCH TANK ASH
5 INCINERATOR
1 (COMPUTER ROOM) DATA ACQUISITION SYSTEM

When compared to the old design and its emission characteristics the modified (state-of-the-art) process proved to be a very substantial improvement. Perhaps of greatest significance was the improvement the modifications had on the particulate generation and characteristics. Since most parameters of concern are associated with the particulates, any improvement in particulate emission rate has a major impact on overall emissions. Figure VIII shows the improvement combustion changes above have made in particulates. Figure IX shows how this improvement typically translates to other parameters such as PCDD. Overall, the combustion modifications have reduced directly, or otherwise PCDD emissions and others, by an order of magnitude. In addition, it is clearly established that "good" operating conditions are a significant factor in terms of emissions as well as energy generation.

Figure VIII - Particulate Emission Comparison

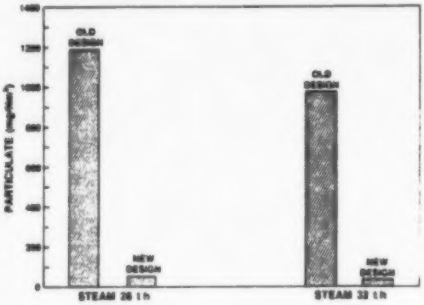
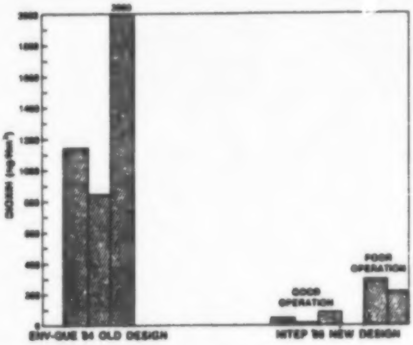


Figure IX - Dioxin Emission Comparison



REFUSE DERIVED FUEL

The refuse derived fuel process has been tested under NITEP in a cooperative test with the US Environmental Protection Agency. The results of these tests will be reported later in this forum and in the spring of 1990 in report form.

CONTROL MEASURES

In Sweden attempts have been made to identify sources of contaminants in MSW (Figures X and XI). Early attempts of this are familiar to most people who are concerned about the excess of mercury in the environment. Initiatives at the consumer level have been successful in further reducing mercury emissions from MSW incinerators by 50%.

Figure X - Heavy Metal Content of Swedish MSW
% Metal in Waste Fraction

	Co	Cu	Ni	Zn
Plastics	1	2	1	1
Paper	5	11	3	11
Animal Matter	1	1	1	1
Vegetable Matter	1	3	3	4
Textiles	1	2	1	1
Rubber and Leather	1	1	1	9
Metal Waste	88	22	87	64
Others	4	63	6	8

FIGURE XI - Heavy Metal Content of Swedish MSW
% Metal in Waste Fraction

	Cd	Cr	Hg	Pb
Plastics	26	5	10	5
Paper	4	7	13	3
Animal Matter	1	1	2	1
Vegetable Matter	2	2	6	2
Textiles	1	1	4	1
Rubber and Leather	4	42	3	2
Metal Waste	60	43	60	85
Others	3	3	3	4

Control programs, both current and proposed, for Sweden are shown in Figures XII and XIII. Long acknowledged as leaders in this area, Sweden is challenging the technology in a realistic manner. The objectives established and proposed are consistent with demonstrated combustion and emission control technologies.

Figure XII - Emissions from MSW Incinerators in Sweden
(mg/nm^3 dry gas 10% CO)$_2$

Substance	Present situation	After measures
Particulates	50	20*
Hg	0.4	0.08
Cd	0.05	0.02
Ni	0.002	0.001
Cu	0.2	0.06
Pb	3	0.5

* calculated as a monthly average

Figure XIII - Emissions from MSW Incinerators in Sweden
(mg/nm^3 dry gas 10% CO)$_2$

Substance	Present situation	After measures
HCl	1000	100*
HF	10	1
SO_x	400	200
NO_x	400	400
PAH	100	0.1
Dioxins, TCDD equiv.	25	0.1-2.0**

* calculated as a monthly average
** 0.5-2.0 for existing incinerator (ng/nm^3)
0.1 for new incinerator (ng/nm^3)

THE CANADIAN PROPOSAL

In Canada, NITEP has gathered sufficient data that, when considered in the global data set, has allowed Environment Canada to make a series of proposals. These proposals have been developed through a Federal/Provincial quasi-government organization called the Canadian Council of Resource and Environment Ministers (CCREM). This organization has developed a national guideline for emissions from MSW incineration. The premise of the proposal is that it is based upon the application of best available control technology (BACT) in combination with more restrictive requirements should local air quality objectives so dictate. The basic proposal recommends a series of design and operating criteria as shown in Figure XIV. The parameters which are recommended for regulation are particulate matter, hydrogen chloride, carbon monoxide and PCDD/PCDF as shown in Figure XV.

Figure XIV - Design Guidelines for Municipal Solid Waste Incinerators

	Combustion Technology			
	Modular			
Parameter	Starved Air	Excess Air	Mass Burn	Refuse Derived Fuel
Minimum Temperature	1000°C	1000°C	1000°C	1000°C
Minimum Retention Time	1 second after final secondary air injection port	1 second after final secondary air injection port	1 second at 1 metre above last air injection ports	1 second at 1 metre above last air injection ports
Primary Air Supply	multi-port injection	multi-port injection	multiple plenums with individual air flow control	air distribution matched to waste distribution
Secondary Air Capacity	up to 80% of total air required	as required by design	at least 40% of total air required	at least 40% of total air required
Auxiliary Burner Size	60% of total rated heat capacity	60% of total rated heat capacity	60% of total rated heat capacity	60% of total rated heat capacity
Oxygen at Boiler Outlet	6 to 12%	6 to 12%	6 to 12%	3 to 9%
Turndown Restrictions	80 to 110% of design capacity	80 to 110% of design capacity	80 to 110% of design capacity	80 to 110% of design capacity
Maximum Carbon Monoxide (4-h rolling average)	50 ppmdv (57 mg/Rm^3) @ 11% O_2	50 ppmdv (57 mg/Rm^3) @ 11% O_2	50 ppmdv (57 mg/Rm^3) @ 11% O_2	100 ppmdv (114 mg/Rm^3) @ 11% O_2

Note: These are guidelines only; incinerator systems with operating parameters outside these ranges that meet the stack discharge limits would be acceptable.

Figure XV - Stack Discharge Limits (@ 11% O_2)

Contaminant	Limit
Particulate Matter	20 mg/Rm^3
Hydrogen Chloride (HCI)	75 mg/Rm^3* (50 ppmdv) or 90% removal
Carbon Monoxide (CO)	57 mg/Rm^3** (50 ppmdv)
Total polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans	0.5 ng/Rm^3*** (Toxic Equivalency Factor new International Method)

* the less restrictive of these requirements shall apply

** RDF systems shall maintain a limit of 114 mg/Rm^3 (100 ppmdv)

*** based upon isomer specific analytical test data; however, if only homologue test data are available, then the most conservative (largest) equivalency factor should be applied.

If basic design recommendations, including an emission control operating temperature of 140°C are met, along with the regulatory limits, then the emission levels shown in figure XVI can be expected to be met for a conventional MSW feed.

Figure XVI - Anticipated Emissions from MSW Incinerators Under Good Combustion Conditions with Dry Scrubber/Fabric Filter Systems

Contaminant	Typical Concentrations*
SO_2	260 mg/Rm³ 100 ppmdv
NO_x	400 mg/Rm³ 210 ppmdv
Pb	50 ug/Rm³
Cd	100 ug/Rm³
Hg	200 ug/Rm³
As	1 ug/Rm³
Cr	10 ug/Rm³
PAH	5 ug/Rm³
PCB	1 ug/Rm³
CP	1 ug/Rm³
CB	1 ug/Rm³

*Concentration at 11% O_2.

GLOBAL TRENDS

Figures XVII to XX compare requirements in European jurisdictions for the parameters of particulate matter, hydrogen chloride, carbon monoxide and sulfur dioxide. Of particular note are the European Economic Community objectives which, if more stringent than a given domestic European criteria, must be enforced. The Canadian proposals which are being incorporated into provincial and federal regulatory instruments are well within the norm.

Figure XVII - MSW Incinerator Particulate Matter Emission Limits (mg/Rm^3 @ 11% O_2)

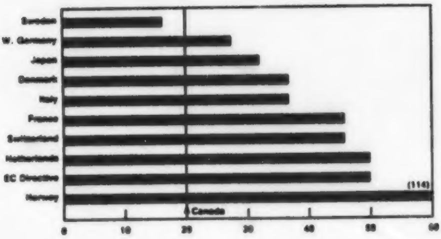


Figure XVIII - MSW Incinerator HCI Emission Limits
(mg/Rm^3 @ 11% O_2)

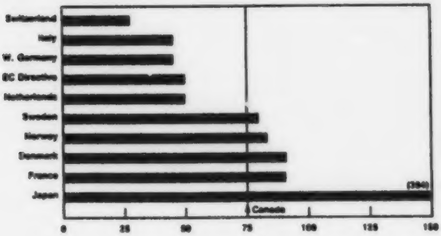


Figure XIX - MSW Incinerator Carbon Monoxide
Emission Limits
(mg/Rm^3 @ 11% O_2)

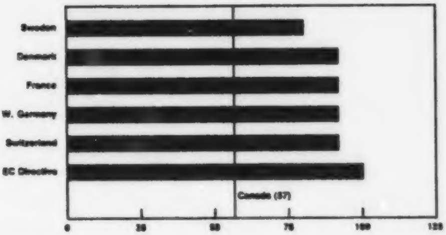
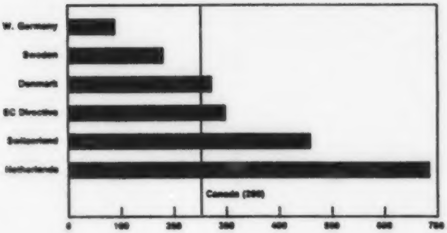


Figure XX - MSW Incinerator
SO_2 Emission Limits
(mg/Rm^3 @ 11% O_2)



The tendency, however, is for continually decreasing values for emission parameters. The public expectation of zero discharge or virtual elimination must be recognized and respected and delivered within the bounds of technological achievement.

OTHER CONSIDERATIONS

For MSW incinerators there are three issues beyond the basic criteria needed for the design and operation of new systems. One is the recognition of the need to deal with existing systems. The second recognizes the "small" system needs and the third addresses the issues associated with MSW incinerator ash residue management.

In Canada we have stipulated that existing facilities must in time comply with the specified emission criteria. Each province will determine its own time frame. Federally, the proposal is 3 years.

For small systems, recognition is given to the fact that remote communities with little opportunity for significant alternative waste management options and those with poor topographical/geological and transportation circumstances must be accommodated. Some modification to the basic emission criteria may be given but basic design and operating conditions must still be met. Few exemption criteria are proposed for federal institutions.

The ash question will be discussed at length by others in this conference. The recommendations in the National Guidelines provide for the disposal of bottom ash in conventional municipal landfill under the stipulation that bottom ash and fly ash are collected and disposed of separately. Fly ash, on the other hand, is not recommended for disposal in conventional municipal landfills and should receive some form of treatment prior to disposal. The primary concern is one of metals mobility.

This latter question of ash residue management remains as something which each jurisdiction is going to have to deal with in the context of its own legislation. As a final remark I would suggest that pass/fail tests established in relation to a specific statute do not necessarily provide the best guidance for the management of this material. A battery of tests should be required so that we are able to assign a waste material a position within our legal framework, but also, hopefully, define through the information gained the "best" management option for its ultimate disposal. This is the avenue we hope to pursue in Environment Canada and which I recommend to others.

U. S. REGULATORY, RESEARCH, AND LEGISLATIVE ACTIVITIES RELATED TO MWC FACILITIES

by

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ABSTRACT

The U.S. Environmental Protection Agency is currently developing air pollution emission rules for new and existing municipal waste combustion (MWC) facilities pursuant to Section 111 of the Clean Air Act. These rules are to be proposed in November 1989 and promulgated in December 1990. This paper provides information on EPA's activities related to this regulatory effort. These activities include assessments of combustion and flue gas cleaning technologies employed at MWC facilities; the collection and evaluation of MWC air pollution emission test data; the performance of EPA-sponsored tests at five MWC facilities; the development of recommendations for good combustion practices at MWC facilities; and an evaluation of the performance and costs of employing alternative combustion and flue gas cleaning strategies for controlling air emissions at MWC facilities.

A major issue in the U.S. is the leaching of trace elements from MWC residues. Work to characterize residue properties and evaluate stabilization techniques is briefly discussed.

Proposed legislation is before the 101st Congress (S. 196) to control air pollutant emissions from municipal waste incineration and provide for safe disposal of ash. Key provisions of this proposed legislation are summarized.

RDF PROCESSING PLANT MAINTENANCE MANAGEMENT

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ABSTRACT

The performance of Refuse Derived Fuel (RDF) processing plants has improved in the last 10 years. However, the maintenance requirements of RDF processing systems has received little formal attention. The result has been that the newer plants have received ample planning from the production viewpoint. Most of the technical work deals with the production side of the issue: designing or manufacturing the equipment or system to work more efficiently or at greater throughput. But, as in other industries, the maintenance perspective may not be included in its complete range of needs; rather it is given to experienced craftsmen. The experienced craftsman needs much more management and staff support to operate productively.

This paper surveys maintenance management articles primarily from the pulp and paper industry and illustrates maintenance management organization, work orders, training, designing out maintenance, and contract maintenance as they apply to RDF plants.

MAINTENANCE MANAGEMENT DEFINED

Maintenance management is defined herein as those management activities required to keep a facility in an as-built condition, continuing to have its original productive capacity (1). For purposes of this discussion, it is assumed that the goal of an RDF processing facility is to provide a contract quantity and quality of RDF.

RDF processing is a process industry in the same way that the pulp and paper industry is a process industry wherein a unit process is preceded and succeeded by another process (1). Concepts primarily from the pulp and paper industry are examined because the maintenance management information in the RDF field is very sketchy. The woodyards in a pulp mill include the processes similar to RDF processing in that sizing, screening, conveying, storage and reclaim are used on an equivalent scale. The digestion, bleaching, paper making and chemical recovery processes are more complex processes and probably are the drivers that force the pulp and paper industry to utilize good maintenance management.

It is acknowledged that RDF processing, indeed the entire municipal waste combustion field, has its own special technical and management problems. But the purpose in this paper is to focus on maintenance management issues.

NEED FOR MAINTENANCE MANAGEMENT

Maintenance management is an important topic to the RDF processing plant for several reasons. First of all, new, capital-intensive RDF plants are dependent upon maintenance management to support the intended performance. Secondly, the RDF plant, whether municipally or privately operated, provides an essential public works service of waste volume reduction and must operate reliably.

Present Conditions

New RDF facilities will cost from $50 to $200 million at an approximate cost of $100,000 per daily ton of capacity, including a power plant. This value is 10 to 20 times what it was 20 years ago.

The plants are complex. Computers have worked their way into RDF plants, and all of the equipment items have become more sophisticated, both mechanically and from the instrumentation and control standpoint. This same trend has occurred in other process industries. This trend accounts for some of the increased cost. Stricter air pollution control standards have forced the use of dry scrubbers, baghouses and de nox systems in some localities.

The actual performance has not necessarily reflected the increased sophistication. A report of an RDF plant that started up in 1983, stated that serious problems were encountered with the RDF fuel and ash systems, and RDF surge bins required high maintenance (2). Another RDF plant that started up in 1981, stated that the design goal of 1000 TPD was not reached due to operating and maintenance problems (3). After modifications, an average test rate of 656 TPD was attained.[1] Since 1983, at least sixteen RDF plants with an aggregate capacity of 22,710 TPD have been built or are under construction (4). Results of early operations of these later plants are not yet widely reported, if at all, in the technical journals.

How much of this shortfall in throughput occurred because the maintenance viewpoint was inadequately represented during planning? The information is sketchy with regard to RDF plants. But it is better known in the pulp and paper industry. For example, at a paper mill in Florida, the Superintendent of the 150-man maintenance department indicated that he had had production problems relating to maintenance (5). Capital improvements had been completed that raised production from 825 tons per day to 1700 tons per day of product. (At today's paper prices, that is equivalent to $765,000/day or $240 million per year of product). No new operators were added because the mill was automated. But also, no new maintenance staff was added. The maintenance department found that it had to work 60 to 80 hours per week, and even then, the essential maintenance could not always be completed. Output was jeopardized. At a cost of $765,000 per day, there had to be a prompt analysis and solution to the problem. The problem was defined as follows: an equipment problem, inadequate staffing or poor training.

At this particular plant, the maintenance staffing and training were found to be inadequate. And they were inadequate because the mill management was so involved in production that they simply forgot that the doubling of the plant equipment greatly increased the load on the maintenance staff. New staffing and training resolved the problem.

RDF plants have had their share of equipment problem, but it is possible that, like the paper mill, the exclusion of the maintenance perspective during capital development, inadequate maintenance staffing and poor maintenance training has contributed to reduced throughput.

Productivity depends upon maintenance as much as upon operations. The reasons for this are severalfold. New plants in general are utilizing more advanced technology. Although the RDF equipment consists of shafts, bearings, chains, sprockets and gears, the control equipment is more sophisticated as it is in industry at large. This move toward more complex equipment requires a great deal more training of maintenance personnel than of operators.

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[1] Frequently more than 1000 TPD was put through the plant under routine operating conditions.

Future Conditions

Maintenance expense is projected to continue increasing as a percentage of product cost, (16) and the maintenance department will become primarily responsible for plant performance. The future plant may have 70 to 80 percent of the workers in the maintenance staff (8).

WAYS TO IMPROVE MAINTENANCE MANAGEMENT

Several schemes to remedy the situation include organizing for maintenance management to maximize plant performance. Another approach would be to build simpler, cheaper plants, but this seems unlikely in the face of environmental pressures that make public issues even of traces of such substances as dioxins.

Another method is the Total Productive Maintenance or TPM developed in Japan. At present, this approach seems most likely to be adopted in an RDF plant after successful implementation in a pulp and paper mill or other process industry.

PROPOSED IMPROVEMENT: STRUCTURE THE ORGANIZATION TO OPTIMIZE THE MAINTENANCE CAPABILITY

The maintenance management system will include the following in varying degrees of emphasis (1):

-Organization for maintenance
-Scheduling maintenance work
-Preventive maintenance (PM)
-Management and supervision
-Work order system and paper flow control
-Time and work reporting
-Priority guidelines
-Backlog information
-Information system, probably computerized
-Inventory control
-Work measurement / work standards

The organization, work order and paper flow control, PM, training, contract maintenance, measurement, and designing out maintenance are dealt with below. Other matters such as work standards, work measurement, computerized information systems and others are not discussed because of space and time limitations. However, each item is essential in a complete maintenance management system.

ORGANIZATION

Idhammer illustrates some plant organizations with the following illustration (6): "Picture the maintenance manager driving his car blindfolded. Beside him sits the mill manager who is peering in the rearview mirror. In the back seat sits the production manager urging the driver to go faster, at the same time warning him about a flat tire." This arrangement won't last long before the car comes to a stop, one way or another.

Top management is responsible for avoiding this kind of situation. It must set the tone of the maintenance management program (15). Top management must realize that complex, integrated and automated plants have good maintenance management as an integral part of the RDF production program. Management must see to it that the maintenance manager has the budgetary resources to hire the necessary specialists to support a work order system, a preventive maintenance program, a training program, and to provide maintenance planning.

An example of a forward looking maintenance organization shown in Figure 1. This organization is based on a modern pulp and paper mill in the Southeast United States (7). The maintenance manager reports at the same level as the production manager. The plant engineering function reports to the maintenance manager - the reverse of the traditional U.S. hardgoods manufacturing. A maintenance planner has been assigned responsibility for each major equipment item or mill area. Many of the maintenance planners are degreed engineers, as are many of the mill engineers. Of the total 99-person maintenance, engineering and foreman staff, two-thirds are white collar positions, of which 14 percent are maintenance planners. The maintenance staff participates in plant capital improvement projects. This mill is successful, having been expanded twice and now undergoing its third major expansion program to triple its production rate since the early 1970's.

WORK ORDER SYSTEM AND PAPER FLOW CONTROL

The Work Order

The heart of a maintenance function is the work order. Work orders provide the written instructions that direct resources to a job in the plant.

The person who authorizes the work order should be aware of the overall maintenance strategy, production realities, practical matters of equipment maintenance and the engineering intent of the system design. He or she should also be in close communication with the plant manager. Work orders should focus on important tasks to the exclusion of unimportant tasks.

A work order system helps to increase the percent of scheduled maintenance. Scheduled maintenance is a more efficient use of time than break-in maintenance because there is less wasted time. Wasted time is time spent looking for tools, waiting for parts, gathering together the correct crew, searching for technical documentation, and hunting part numbers. Wasted time may be as high as 70 percent during break-in maintenance (6). Factors affecting wasted time are skills, flexibility in the maintenance organization, ease of finding spare parts when only the equipment number is known, and access to technical data. The use of a work order system provides a method whereby these items are resolved in advance of the field work.

Equipment, manpower, materials, purchasing, maintenance planning and engineering, equipment records and preventive maintenance data are all affected or derive important information at some point in the life of the work order.

As can be seen in Figure 2, an example workorder system is depicted (1). The work order requests may be initiated by foremen, operators, engineers and craftsmen. The maintenance engineer provides the technical detail necessary to describe the problem and the approach to fixing it.

A maintenance planner provides the equipment coding, the parts and materials specifications, verifies their availability in stores or issues instructions to purchasing. The maintenance planner adds a priority to the work order and establishes the labor time required based on standards or plant experience, schedules tools, and obtains correct authorization from the maintenance manager. The work order is transmitted to a foreman.

Knowing that the materials, parts, tools, manpower and technical data all are available, the craftsman can proceed with the job. Upon job completion, the work order is used to record the time, materials, and equipment used. Comments on conditions by the foreman can be noted. The form is transmitted back to a file and to maintenance planning for record purposes.

As appropriate, real time networked computer systems, individual computer stations or hand written work orders can be used so long as the completed work order and job completion data are accurately recorded and transmitted in permanent form to file.

Paper Flow and Data Control

Paper flow control is important because the work order records both the decision to commit resources to a job and is a record of the results of that decision. Sporadic paper flow control and lost or delayed work orders frustrate the ability of the maintenance manager to implement joint production maintenance plans. Therefore, the system of checking the original work order against the completed work order and reasons for delay should be provided.

The data contained on the completed work order can be valuable information if used properly. It is a permanent record of what has happened to a particular piece of equipment at a point in time. When a bearing seizes, the first question might be when was it last replaced. If a written record is maintained, the fact that the bearing has failed 5 times in 5 months will be an indicator to the maintenance planner that the bearing may not be the root problem. Without a written record, the repair may continue without really getting to the bottom of the problem.

A written record makes permanent what otherwise would be temporary, probably verbal data: a phone call or face-to-face request. The fact that a new bearing seized probably would seem commonplace and therefore of little value to a craftsman. And he may or may not report it. But when reduced to writing, what might have been casual conversation becomes a potentially powerful statistical data base in the hands of a PM specialist analyzing bearing life.

The written record of time spent doing specific jobs provides good time standards for the maintenance planner. The use of time standards also enables the measurement of maintenance work. The existing time standards for well defined tasks are the Universal Maintenance Standards (UMS) and the Maintenance Time Methods (MTM). Normally, these are used to establish time standards for typical jobs such as replacing a bearing or shaft. Because RDF plant jobs may not be comparable, RDF time standards can be established by the maintenance planner using RDF plant job time records.

Work standards will help to create clear expectations of performance and help instill discipline in the overall maintenance system.

PREVENTIVE MAINTENANCE (PM)

PM is maintenance that occurs before the RDF plant has reduced quality or quantity of throughput. The PM specialist in today's complex plant is of central importance because downtime and break-in maintenance is prohibitively expensive. The PM specialist can reduce these expenses. By utilizing observed physical parameters such as temperature, vibration, oil analysis, electric resistance, and shock pulse measurements the specialist can quantify the condition with objective data. This data will provide strong incontestable evidence to the production manager that maintenance must be performed.

The specialist must keep abreast of the latest diagnostics technologies. These include automated vibration analysis, stroboscopes for coupling wear, lasers for alignment, and eddy current testing for thickness measurement and to find cracks. These technologies plus a dedicated PM specialist will frequently enable prediction of outages. And the outage can then be avoided in a timely manner.

During investigations for plant betterment, one RDF plant found that acceptable RDF was produced during the early hours of shredder operation with new knife blades. However, it was necessary to watch the particle size closely because it tended to deteriorate all at once after a certain number of hours of operation (2). If this had been a routine operation, a PM specialist would predict the allowable operating time for the knife blades, and schedule replacement just prior to the RDF deterioration.

The result of vibration analysis based on horsepower in an Esso Petroleum Canada plant saved a large amount of money. It was possible because the PM specialist had enough data to recognize that while the annual cost of maintaining small motors was much smaller than large motors on a dollar per horsepower basis, the aggregate horsepower of small motors so greatly exceeded that of large motors, that simply concentrating on timely PM of small motors would save money. The plant showed a reduction in time between repair of pumps from 15 months in 1984 to 25 months in 1987 and in some cases 48 months. Costs per day for rotating equipment for that plant in January 1987 were $3700 to $4000. By September 1988, it had been reduced to $1500 per day. Plant wide, that was a reduction from $1.3 million per year to $528,000 per year--a savings of $744,400 per year (8).

A refinery is not an RDF plant. But RDF plants do have more small motors than large motors. And the fact that significant savings were possible in a mature process industry, significant potential savings may exist in an RDF plant. In a paper mill, there will usually be a 1% to 2% increase in "prime quality production" and a 20% reduction in maintenance costs when the right PM is implemented (6). For RDF, this kind of performance increase has not been quantified. Usually, the greatest percentage increase in performance occurs in evoloving technologies; because RDF is an evolving technology, most will agree, there may be considerable potential of increased performance. Whatever potential exist, it can best be realized by systematic PM using the latest technology.

TRAINING

Training programs for a maintenance staff should ideally be both broad and detailed. A career advancement line "of progression" should be established at the outset so that a maintenance worker gains the right kind of experience (5). Included in this progression should be a significant stint on the production side of the plant. This experience will be valuable, perhaps essential later, in establishing work order priorities.

The proper procedures for inspection, measurement, dissasembly, inspection, testing, reassembly, and cleanup including safety procedures, job reporting, time reporting and other related tasks must be well defined by the maintenance manager. The number of RDF plants in the U.S. and the rest of the world is too small to expect a widely available RDF-specialized labor pool. The only way an RDF plant can assure that its maintenance staff is properly trained is to train the staff over and above the generic skills of the craftsman, PM specialist, and foreman.

A line of progression that includes maintenance responsibility for different areas of the plant is desirable. The progression might include responsibility for the stores area, the conveyors, the screening systems, the shredders, the tipping and unloading areas and so forth. Foremen and superintendents, maintenance planners, PM specialists, maintenance managers and plant managers have to be drawn from the ranks, and they should have a well rounded experience.

CONTRACT MAINTENANCE

Owners/operators sometime contract the operation and maintenance of RDF facilities. Contract maintenance will have advantages if the plant management exercises its inherent responsibility of providing leadership, by setting the tone of the contract operation and by management that is knowledgeable in the maintenance field (1).

But the contractor must also bring specific abilities to the plant organization including management support, technical support, training capability, parts support, operations support and applications and engineering support (1). In a paper mill in the State of Texas, the mill management and the contract maintenance work so closely together that the mill employee is indistinguishable from the contractor's employee, except of course by the name of their employer (5,9,10). In such instances, contract maintenance may be the most cost effective.

MEASUREMENT OF MAINTENANCE

Numerous maintenance measurements are available. The information used to apply a method will come largely from plant work orders and other records. Because a person is being measured on the very information he or she is entrusted to supply, that information should always be used in a positive way by the plant management. Otherwise, the source of valuable maintenance information will dry up.

The cost of maintenance will probably always be looked at by management. But to simply look at one number as an expense item does not do it justice. One plant manager reported that he could increase plant performance measurably by spending more on maintenance, and that it was easy to show that such maintenance more that paid for itself. However, whenever he presented maintenance as a line item in his budget to the Corporate Vice President, it was crossed out. His maintenance budget had to be embedded in his production budget (7).

So, the maintenance cost should be measured with respect to plant production to put it into perspective. The expenditure can be related to the production performance as seen in Table 1. Another measure of maintenance is equipment availability that indicates whether the equipment is working properly.

Many other measurement methods are available to suit individual needs of a plant. A positive approach that looks at maintenance as an integral factor in plant production will yield the most effective maintenance operation.

TABLE 1. MEASUREMENT OF MAINTENANCE COSTS

$$\frac{\text{Maintenance Cost in Dollars}}{\text{Throughput in Tons}} = \$/\text{Ton}$$

$$\frac{\text{Equipment Running Time} \times 100}{\text{Run Time} + \text{Down Time}} = \%$$

DESIGNING OUT MAINTENANCE

The best preventive maintenance dollar is spent during the design stage by close attention to detail and good system design. A PM specialist, maintenance engineer, maintenance planner, and foreman can eliminate system design faults before they are constructed. At one plant, the maintenance planners routinely attend capital project planning meetings with consultants and vendors for this specific purpose. A change that could be made early in the design stage by maintenance planners would cost 7000 times more in terms of life cycle maintenance costs if it were done after the project was constructed (6).

But just as importantly, the maintenance manager must insure that the plant incorporates the necessary features that are required for effective maintenance. Access is one of the most important of these features. The stores room, the computer system, and maintenance technologies may be installed with a new plant or later as capital improvements. The ideal to aim for when designing out maintenance would be like a household refrigerator that may last 20 years without any maintenance.

Expansion or improvement projects are a second area where the maintenance manager can design out maintenance. He should have lists of necessary plant improvements for pumps, lowering devices, new catwalks to improve access, better equipment specifications. These lists should be accumulated as input from the maintenance planner, the PM specialist and others who can point out the frequent and the important items, not just the rare and inconvenient one that has little or no effect on plant production.

THE DESIRABILITY OF IMPROVED MAINTENANCE MANAGEMENT

The biggest return on the RDF plant maintenance dollar is from planned maintenance including involvement of those trained in systems design in capital projects from the maintenance viewpoint.

If the trends in the pulp and paper industry are any guide, then the cost of RDF plant maintenance can be held steady even with increased production when modern maintenance management is implemented. RDF plant production could be increased with a minimal increase in operating payroll. At the same time the essential public service of refuse volume reduction could be maintained without unnecessary delays or costs to the public.

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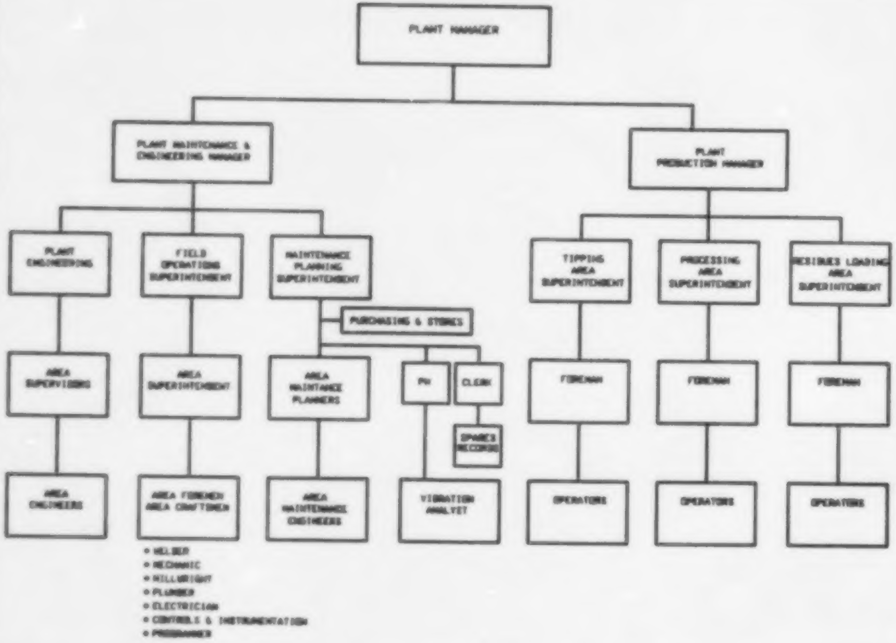
PLANT MANAGER
PLANT MAINTENANCE & ENGINEERING MANAGER
PLANT PRODUCTION MANAGER
PLANT ENGINEERING
FIELD OPERATIONS SUPERINTENDENT
MAINTENANCE PLANNING SUPERINTENDENT
PURCHASING & STORES
TIPPING AREA SUPERINTENDENT
PROCESSING AREA SUPERINTENDENT
RESIDUES LOADING AREA SUPERINTENDENT
AREA SUPERVISORS
AREA SUPERINTENDENT
AREA MAINTANCE PLANNERS
PM
CLERK
SPARES RECORDS
FOREMAN
FOREMAN
FOREMAN
AREA ENGINEERS
AREA FOREMEN AREA CRAFTSMEN
AREA MAINTENANCE ENGINEERS
VIBRATION ANALYST
OPERATORS
OPERATORS
OPERATORS
• WELDER
• MECHANIC
• MILLWRIGHT
• PLUMBER
• ELECTRICIAN
• CONTROLS & INSTRUMENTATION
• PROGRAMMER

Figure 1. Plant Organization For Emphasizing Maintenance

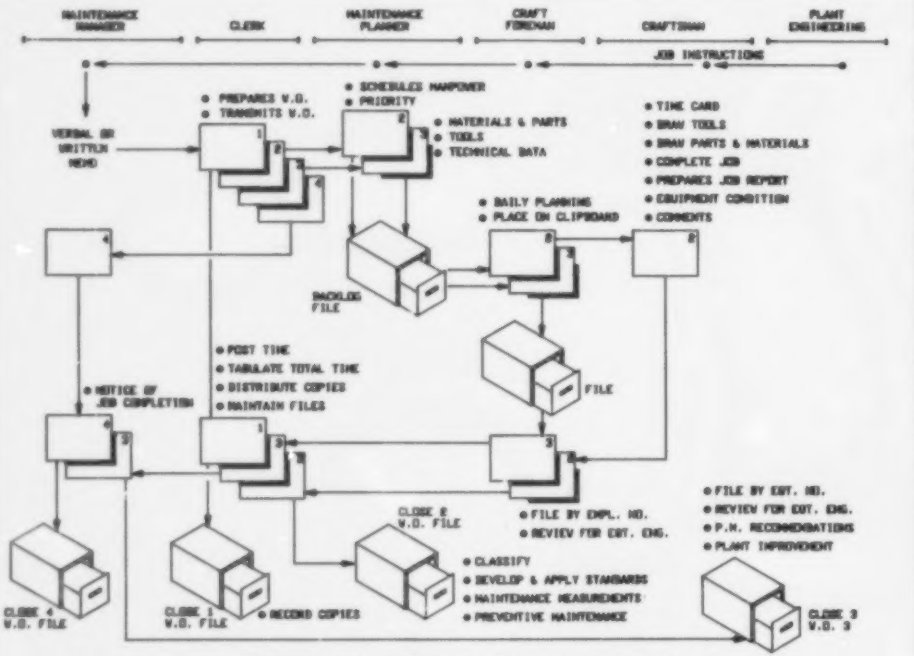
MAINTENANCE MANAGER
CLERK
MAINTENANCE PLANNER
CRAFT FOREMAN
CRAFTSMAN
PLANT ENGINEERING
JOB INSTRUCTIONS
VERBAL OR WRITTEN MEMO
● PREPARES W.O.
● TRANSMITS W.O.
● SCHEDULES MANPOWER
● PRIORITY
● MATERIALS & PARTS
● TOOLS
● TECHNICAL DATA
● TIME CARD
● DRAW TOOLS
● DRAW PARTS & MATERIALS
● COMPLETE JOB
● PREPARES JOB REPORT
● EQUIPMENT CONDITION
● COMMENTS
● DAILY PLANNING
● PLACE ON CLIPBOARD
BACKLOG FILE
FILE
● NOTICE OF JOB COMPLETION
● POST TIME
● TABULATE TOTAL TIME
● DISTRIBUTE COPIES
● MAINTAIN FILES
● FILE BY EMPL. NO.
● REVIEW FOR EQT. ENG.
● FILE BY EQT. NO.
● REVIEW FOR EQT. ENG.
● P.M. RECOMMENDATIONS
● PLANT IMPROVEMENT
CLOSE 2 W.O. FILE
● CLASSIFY
● DEVELOP & APPLY STANDARDS
● MAINTENANCE MEASUREMENTS
● PREVENTIVE MAINTENANCE
CLOSE 4 W.O. FILE
CLOSE 1 W.O. FILE
● RECORD COPIES
CLOSE 3 W.O. 3
1
2
3
4

Figure 2. Work Order & Flow Control (1)

COMPREHENSIVE WASTE MANAGEMENT STRATEGIES INCORPORATING INCINERATION

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ABSTRACT

There is no one specific method of waste management that is a panacea for society's needs.

Working actively with the private sector, the Region of Hamilton-Wentworth participates in the successful and effective processing of approximately 600,000 tons of solid wastes annually.

In a recent initiative, the Region commissioned the development of new strategies to maximize incineration with energy recovery, and materials reclamation, while minimizing its landfilling requirements. These staged strategies, which will reduce landfill demand at the Region's disposal site from 250,000 to approximately 66,000 tons per year, are described in this paper.

INTRODUCTION

Incineration of municipal solid waste has been practised since the early 1900's. Volume reduction of municipal solid waste, of greater than 90%, is achieved by incineration. The early incinerators consisted of small excess air retractory furnaces and did not employ energy recovery or air emission control devices.

Designs of incinerators gradually improved with significant development occurring in the 1950's when multi-chamber and controlled air incinerators were developed.

In the 1960's and 1970's, incinerators were built with energy recovery and air emissions control equipment, mostly electrostatic precipitators.

In the 1980's, impressive improvements to incineration technology were made in the following areas:

- combustion air distribution
- boiler design to minimize flame impingement on tubes and to reduce deposit formation on tubes
- process control (programmable logic controllers)
- air emissions control with lime injection and fabric filters.

In countries such as Denmark, Switzerland, Sweden and Japan, incineration of municipal solid waste is a major component in their waste management programs with close to 50% of municipal solid waste being converted into usable energy.

In North America, incineration is a minor component of waste management with only about 5-10% of municipal solid waste being converted into usable energy. Incorporation of modern incineration as a component of waste management is becoming more common. The three most prevalent incineration methods used are mass burning, two-stage combustion and semi-suspension burning.

Currently, in North America, more emphasis is being placed on balanced and integrated waste management programs in which incineration with energy recovery is an important component but not necessarily the major component. Processing of municipal solid waste prior to incineration is becoming the trend, to increase recycling and to improve the quality of municipal solid waste as a fuel.

The disposal of ash residues from incinerators has recently become a concern and significant development activities are underway for treatment of the ashes.

The advantages of incineration in an integrated waste management program are:

- significant volume reduction (greater than 90%) for non-recyclable waste materials
- cost recovery with sale of energy
- environmental problems can be rectified
- the incineration process can be modified and upgraded to ensure waste is being treated with best available technology.

The disadvantages of incineration are:

- relatively high capital and operating costs
- difficulty for siting.

A progressive waste management system consists of a number of essential components. Recycling, reduction, landfilling and incineration cannot stand alone, or even in limited combinations, in the face of present environmental challenges. Modern and effective waste management practices generally require an integration of most or all of these noted techniques, properly weighted and applied to suit each specific local or regional requirement. The Region of Hamilton-Wentworth in Ontario, Canada has evolved a modern municipal waste management system, which originally incorporated semi-suspension incineration with energy and material recovery features in 1972. This EFW plant, known as SWARU, formed the leading edge for other present day MSW incinerators.

Since 1972, SWARU has been upgraded regularly to keep pace with current standards. It continues to be an integral component of an effective solid waste program which includes curbside recycling, waste exchange, source separation, and household hazardous waste collection, in conjunction with a 1990's landfill/transfer station system.

BACKGROUND

In the early 1960's, municipal councils in the Hamilton-Wentworth area became aware of imminent problems pertaining to solid waste disposal requirements. As a result, several studies were commissioned to evaluate existing conditions and make recommendations for future needs.

In 1968, The Corporation of the City of Hamilton commissioned the design and construction of a municipal incinerator with metal and energy recovery capabilities. SWARU (Solid Waste Reduction Unit) was completed in 1972 at a total cost of $8.95 million.

When the Regional Municipality of Hamilton-Wentworth was formed in 1974, it assumed ownership and operation of solid waste disposal operations, including SWARU. Today, this plant is an integral part of the Region's comprehensive solid waste system, which consists also of a modern landfill site, three fully enclosed transfer stations, and a fleet of transfer trailers. This system serves 6 municipalities with a total population of 414,456 people and an area of 449 square miles. The operation of the Regional Solid Waste Management System was part of a competitive private sector 10-year contract award in 1980.

In today's sphere of waste management, effective programs have evolved from earlier practices of open dumping and burning to systems incorporating some or all of the following techniques:

- approved landfilling measures at existing sites
- new landfills incorporating complete designs for development, operation and closure
- recycling and source separation
- incineration for energy recovery.

Generally, modern-day tactics subscribe to the Ministry of the Environment's 4 R's initiative:

- Reduction
- Re-use
- Recycling
- Recovery.

However, in Ontario, an estimated 10.5 million tons of solid waste are generated annually. Of this amount, 7.5 million tons are buried in municipal landfills and the balance is reprocessed or buried at the generators' facilities.

In Hamilton-Wentworth, each resident generates about three-quarters of a tonne of waste annually. This quantity reflects primarily the garbage collected in the residential/commercial sector, and does not include liquid waste, most industrial waste, or construction and demolition debris. These wastes may equal or exceed those quantities handled in the municipal sector.

In the Province of Ontario, Ministry records indicate the presence of 2,500 documented closed sites and 1,500 active sites. At the present time, approximately 30 applications per year are submitted to the Ministry of the Environment for new landfills, and 80 applications for new facilities are now being evaluated. It is not expected that all applications will be approved.

PRIVATE DISPOSAL FACILITIES

The closure of former landfill sites led to a new economic activity in the Region, the establishment of private disposal facilities. The Region's present system does not accommodate materials such as demolition rubble, excavated materials, and bulk industrial solid wastes. In 1978, before closure of former sites, the Region entered into a co-operative planning process with local private firms to establish new private sites which would receive and process waste materials which do not require the level of protection afforded by the Regional landfill. Valuable landfill space is thereby conserved for the disposal of more appropriate materials.

Under licence by the Ministry of the Environment, a number of firms established disposal sites in old quarries which are approved for landfilling of inert rubble and classified industrial and commercial wastes. This activity diverts a waste stream which would place an unnecessary burden on the costlier Regional disposal system, provides savings to industry in the form of reduced disposal fees, creates additional jobs, and permits rehabilitation of inactive pits and quarries.

The two major private disposal sites in Hamilton-Wentworth are operated by Taro Properties, and Steetley Industries. As a parallel to the Region's disposal system, the local firm of Philip Environmental Services has established transfer stations in the Hamilton industrial core. Also, this company processes various industrial solid wastes for re-use by other industries, while materials with no further commercial value are transported to the outlying quarries for disposal. In a recent development, Philip Environmental established a rural site which processes agricultural and food product wastes for soil enhancement.

The quantity of commercial and industrial non-putrescible wastes disposed of, or processed, by the private sector in the Region is estimated to be at least equal to the 300,000 tonnes per year processed or landfilled by municipal facilities.

SOURCE SEPARATION AND RECYCLING

Third Sector Employment Enterprises is a non-profit agency that was organized in the Hamilton-Wentworth area in early 1978 to deal with two prevalent national problems at the local level:

1. The high rate of unemployment in the 18-25 year old age group; and,
2. The rapid depletion of timber, one of Canada's natural resources.

Established to function as a "private sector" non-profit company, Third Sector was initially subsidized entirely with a "public sector" federal grant.

Prior to 1984, Third Sector dealt exclusively with curbside collection and recycling of newsprint. In accordance with the Region's commitment to resource recovery, Third Sector Enterprises was consulted to develop a multi-material source separation and curbside recycling program. Third Sector's great interest in expanding employment opportunities resulted in the establishment of a separate new venture, Third Sector Recycling.

At present, Third Sector operates on a Region-wide basis and has expanded its operational scope to include the collection and recycling of newsprint, as well as glass and metal.

The local municipalities of the Region and the Ministry of the Environment have supported Third Sector's program by subsidizing the purchase of "Blue Boxes". These boxes are being distributed to all participating householders and serve as a great incentive to promote resource recovery.

In 1988, Third Sector Recycling recovered approximately 13,000 tonnes of newsprint, glass and metal, representing approximately 7% of the total municipal solid waste landfilled annually in the Hamilton-Wentworth Region.

WASTE MANAGEMENT SYSTEM DESCRIPTION

The Regional solid waste management system consists of the following components and operations which currently handle approximately 300,000 tonnes of solid waste annually.

Collection System: required to gather the solid waste at homes or business establishments and deliver it to transfer stations or SWARU. This collection system is the responsibility of the Area Municipalities and private enterprise.

Transfer Stations: strategically situated to effectively serve the disposal needs of the public and waste haulers, these are located at Olympic Drive in Dundas, at Upper Ottawa Street, and at Kenora Avenue in Hamilton. The fully enclosed facilities are depots for receiving and shipping waste which is compacted into large completely enclosed transfer trucks and hauled to the landfill site. The transfer stations provide a means of traffic control, permit improved regulation of the types of waste accepted, and provide an economical point of discharge for the collection agencies. The stations, which began operations in July 1980, can be adapted to front-end processing facilities in a comprehensive resource recovery system. Bulk metals are currently retrieved from the transfer station waste stream for recycling.

SWARU (Solid Waste Reduction Unit): incinerates waste to generate steam and recovers metal. This activity reduces the volume of waste by 95%. SWARU burns municipal refuse in two spreader stoker water-walled boilers, utilizing a semi-suspension combustion process. The plant design capacity is 550 tonnes (600 tons) for a 24-hour day, while the normal daily operation is in the order of 460 tonnes (500 tons) per day. This quantity comprises 1/3 of the total solid waste stream received by Regional facilities, and represents a diversion of almost 91,000 tonnes (100,000 tons) per year that otherwise would be landfilled.

Over the past 14 years, SWARU has received substantial attention to improve materials handling. In 1982, under a joint agreement with the Ontario Energy Corporation, a four megawatt steam turbine generator installation was completed and began operation. Approximately 14 million kilowatt hours of electricity are produced annually and sold to Ontario Hydro.

An additional resource recovery feature of SWARU is the ferrous metal recovery system, which removed approximately 4,000 tonnes of ferrous material from the waste delivered to the plant in 1988.

A $12 million retrofit project was completed in 1986 to improve plant efficiency and emission control.

Haulage System: consists of a fleet of 75 cubic yard transfer trailer vehicles, hauling totally enclosed compacted waste along prescribed routes on major roadways from the transfer stations to the landfill site. The transfer trailers are generally the only vehicles permitted to deliver waste to the landfill site. Traffic impact is minimized and use of the landfill site is optimized. The Region's municipalities also benefit as a result of reduced haulage distances.

Solid Waste Landfill Site: located in the Area Municipality of Glanbrook, where waste is compacted and buried in earth covered cells. This site, which commenced operation in October 1980, was designed and is operated in accordance with the latest standards of the Ministry of the Environment. A continuous and comprehensive environmental monitoring programme is in effect for this facility to provide confirmation of an environmentally secure operation.

System Capacity

The nominal working capacity of the waste management system is 500,000 tonnes (600 000 tons) per year, derived from the following design capacities of the various facilities:

SWARU	100,000 tonnes (120 000 tons)/year
Dundas Transfer Station	119,000 tonnes (130 000 tons)/year
Mountain Transfer Station	119,000 tonnes (130 000 tons)/year
Kenora Transfer Station	319,000 tonnes (350 000 tons)/year

The capacities at the Mountain and Kenora stations can be readily increased by adding additional compactors.

If SWARU is closed for modifications or repairs, the wastes normally delivered there will be detoured to the adjacent Kenora Transfer Station, where it will be processed for disposal at the Regional Landfill Site. This transfer station has sufficient capacity to accommodate the extra diverted waste.

OPTIMIZATION STRATEGY

GENERAL

In March, 1988, WCI Waste Conversion Inc. was commissioned by the Region of Hamilton-Wentworth to evaluate the Regional Solid Waste Management System, and particularly the waste stream quantities and characteristics, to determine the most cost effective way of maximizing deliveries of acceptable combustible waste to SWARU and maximizing the life of the Regional landfill site.

As a result of the work performed, a set of recommendations has been developed, which is presented in condensed form in Table 1.

The principal recommendations of this resource recovery strategy are intended to be implemented in a staged manner, with Phase 1 providing significant reductions in landfill demand at little or no cost to the Region and Phases 2 A and 2 B providing further reductions in landfill requirements and increasing the fuel stream to SWARU, all at relatively low cost. It should be noted that these recommendations can be effected with virtually no impact on the normal waste collection and haulage patterns in the Region, and with little or no disruption to users of the Regional solid waste system.

Phases 1 and 2 can be initiated immediately, with Phase 2 becoming fully functional by 1989. Implementation of Phases 1 and 2 will reduce the annual landfill waste stream from its present level of approximately 250,000 tonnes to 184,000 tonnes.

Phase 3 is a more complex strategy requiring a longer lead time for planning and design, and a substantial capital investment. However, Phase 3 can provide the means of further reducing the annual landfill waste stream from 184,000 tonnes to approximately 66,000 tonnes. A key component of Phase 3 is the generation of upgraded refuse fuel. The proposed compost process employs equipment similar to that which is needed for the upgraded refuse fuel process but of smaller capacity. Experience with the compost process will assist in assessing the equipment for an upgraded refuse fuel process.

EXISTING SYSTEMS

Figure 1 and Table 2 illustrate the existing waste stream flow in the Regional Solid Waste Management System. Discussions with local firms engaged in waste collection and disposal suggest that the additional quantities of waste handled by the private sector outside of the Regional Solid Waste Management System are approximately equal to the size of the waste stream processed by Regional facilities. Also, the waste stream shown in figures contained in this report are exclusive of the curbside collection conducted under the Blue Box recycling program. This quantity was removed at the front end of the waste stream before collection and assimilation into the Regional Solid Waste Management System.

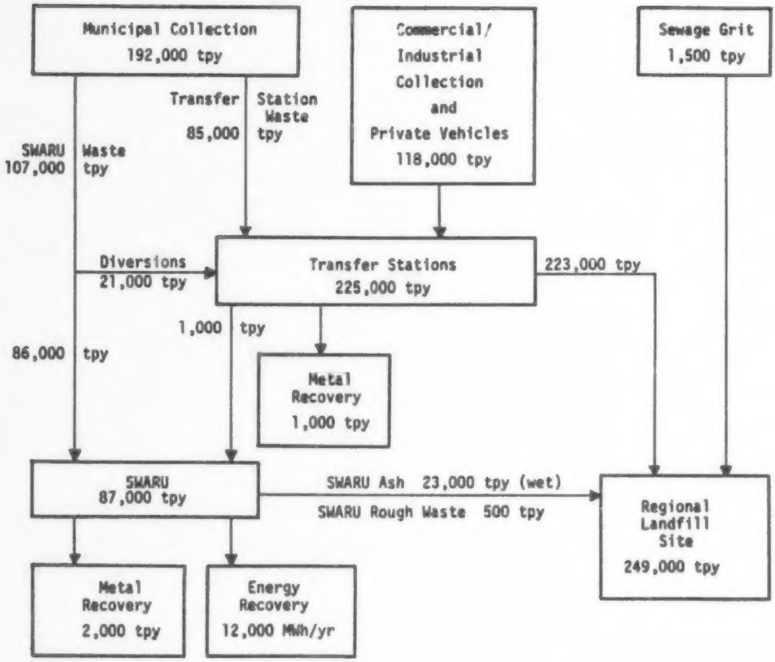
TABLE 1. SUMMARY OF PRINCIPAL RECOMMENDATIONS

RECOMMENDATION	INITIATION	FULLY FUNCTIONAL	ESTIMATED BENEFITS	GROSS ESTIMATED COST TO REGION	MOE FUNDING GUIDELINES
PHASE 1: Waste Stream Control	Immediately	30 days after initiation	-9,000 tpy reduction in landfill -4% site life extension	Nil	N/A
PHASE 2-A: Transfer Station Separation	Immediately	late 1988/early 1989 contingent on budget available	-reduce landfilling by up to 40,000 tpy, increase SWARU process by up to 55,000 tpy -20% site life extension	-$300,000 for transfer station capital works -Tricil annual bonus for SWARU process	50% of capital costs
PHASE 2-B: Corrugated/ Wood Recovery	Immediately	late 1988/ early 1989	-reduce landfilling by 16,000 tpy -recycle 16,000 tonnes of wood and corrugated paper -annual revenue depending upon market conditions -7% site life extension	-$1 million capital works -$250,000/yr operating costs	50% of capital costs
PHASE 3: Fuel Preparation	early 1989	1995 (tentative)	-93,000 tpy of RDF -23,000 tpy of reclaimed materials -site life increased by a factor of 3 in addition to that achieved in Phases 1 and 2 - reduce landfill to 65,000 tpy	-$10 million capital works -$2 million/yr operating costs	50%-100% of capital costs

Quantities are in tonnes unless otherwise specified.

FIGURE 1

Existing Waste Stream Flow


Municipal Collection
192,000 tpy
Commercial/
Industrial
Collection
and
Private Vehicles
118,000 tpy
Sewage Grit
1,500 tpy
Transfer Station Waste
85,000 tpy
SWARU Waste
107,000 tpy
Diversions
21,000 tpy
Transfer Stations
225,000 tpy
223,000 tpy
1,000 tpy
86,000 tpy
Metal
Recovery
1,000 tpy
SWARU
87,000 tpy
SWARU Ash 23,000 tpy (wet)
SWARU Rough Waste 500 tpy
Regional
Landfill
Site
249,000 tpy
Metal
Recovery
2,000 tpy
Energy
Recovery
12,000 MWh/yr

Notes for Figures 1 through 5:

1. Data based upon available 1987-1988 Regional Statistics.
2. tpy = tonnes per year
3. Numerical summaries in the flow charts may vary by ±1% due to minor variances in data provided in records.

TABLE 2 EXISTING REGIONAL WASTE DISPOSITION [a]

1.	Municipal Collection	192,000 tpy
2.	SWARU Waste Stream	108,000 tpy
3.	Process Diversions	21,000 tpy
4.	SWARU Waste From Transfer Stations	1,000 tpy
5.	Actual Waste to SWARU Process (2 - 3 + 4)	87,000 tpy
6.	Municipal Collection to Landfill (1 - 5)	105,000 tpy
7.	Commercial/Industrial Collections and Private Vehicles	118,000 tpy
8.	Sewage Grit	1,500 tpy
9.	SWARU Ash and Rough Waste	23,500 tpy
10.	Total Landfill	249,000 tpy

[a] Based upon 1987 - 1988 Regional data, rounded off to nearest 1,000 tonnes.

WASTE STREAM CONTROL

More consistent enforcement of the Regional Solid Waste By-Law to control the delivery of materials designated as "Non-Acceptable Waste", specifically construction and demolition debris (rubble, long lengths of wood, dry wall, and possibly shingles) would reduce the amount of waste delivered to the transfer stations, and consequently to the Regional Landfill Site.

Benefits

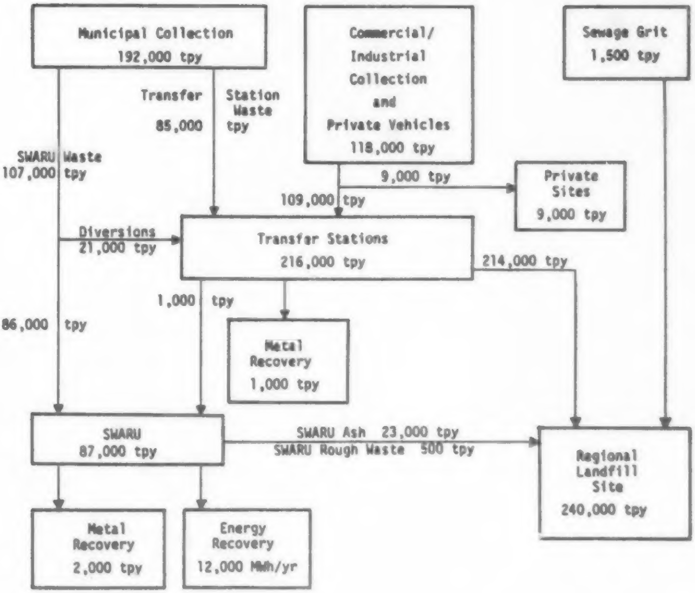
Based upon the field surveys conducted during this study, proper enforcement of the "Non-Acceptable Waste" by-law provision is expected to reduce average waste deliveries to all three transfer stations, and subsequently the landfill site, by an estimated amount of at least 30 tonnes per day. This represents about 9,000 tonnes per year, or 4% of annual disposal at the Regional landfill site. In comparison, the 1988 household "Blue Box" program is expected to divert approximately 13,000 tonnes from landfill. Figure 2 illustrates the materials flow of this recommendation.

Analysis

It is estimated that approximately 55,000 tonnes per year of waste delivered currently to the Dundas and Mountain Transfer Stations would be acceptable at SWARU, as received. This quantity represents the waste available from entire loads delivered at the transfer station which require no processing, sorting, or other removal of materials whatsoever. Therefore, 55,000 tonnes of transfer station waste can be loaded and delivered directly to SWARU for energy and resource recovery with no accessory investment of materials or labour. Because of the noted waste characteristics, it is possible to receive and store identified and designated "SWARU Acceptable Waste" on one half of the transfer station tipping floor, while waste unacceptable for SWARU can be deposited on the other half of the tipping floor for delivery to landfill.

FIGURE 2

Phase 1 Waste Stream Flow:
Improved Waste Control


Municipal Collection
192,000 tpy
Commercial/
Industrial
Collection
and
Private Vehicles
118,000 tpy
Sewage Grit
1,500 tpy
Transfer Station Waste
85,000 tpy
SWARU Waste
107,000 tpy
9,000 tpy
Private
Sites
9,000 tpy
109,000 tpy
Diversions
21,000 tpy
Transfer Stations
216,000 tpy
214,000 tpy
1,000 tpy
86,000 tpy
Metal
Recovery
1,000 tpy
SWARU
87,000 tpy
SWARU Ash 23,000 tpy
SWARU Rough Waste 500 tpy
Regional
Landfill
Site
240,000 tpy
Metal
Recovery
2,000 tpy
Energy
Recovery
12,000 MWh/yr

In actual operation, wastes bound for landfill will be loaded into the transfer trailers continuously as they are received, and shipped out as required. When a full transfer trailer load of SWARU waste is accumulated, it can be loaded and transported directly to SWARU. The improved waste stream control measures detailed in Phase 1 will facilitate the implementation of the Phase 2 strategy. Figure 3 details the waste routing associated with this strategy.

The additional "SWARU Acceptable Waste" generated through the simple floor segregation strategy noted, 55,000 tonnes per year, would increase the total SWARU annual waste stream supply to approximately 141,000 tonnes. (This will reduce the amount of waste to be landfilled by approximately 20%).

An annual process level of 137,000 tonnes is considered to be the maximum reasonable amount that can be processed at SWARU under consistent 6 days per week operation.

WOOD/CORRUGATED PAPER RECOVERY

Benefits

Substantial amounts of wood and corrugated paper being delivered to the transfer stations may be recovered and marketed. This may be done in three ways:

a) Transfer station users may be directed to deposit corrugated paper and wood in designated bins, as is now done for metal scrap.

b) Corrugated-rich loads and waste wood may be directed to a central depot such as Kenora Transfer Station. A simple corrugated paper screening and picking system could be implemented in which the contraries are separated from the corrugated paper in a drum screen.

   The cleaned corrugated paper can then be easily and efficiently hand-picked and baled or compacted for market. The reject material, which could be 30% - 50% by weight can be handled through the existing transfer station system. An extension to the transfer station to enclose the corrugated paper screening, picking and compacting operations would be required. A similar waste wood reduction and processing facility may also be considered.

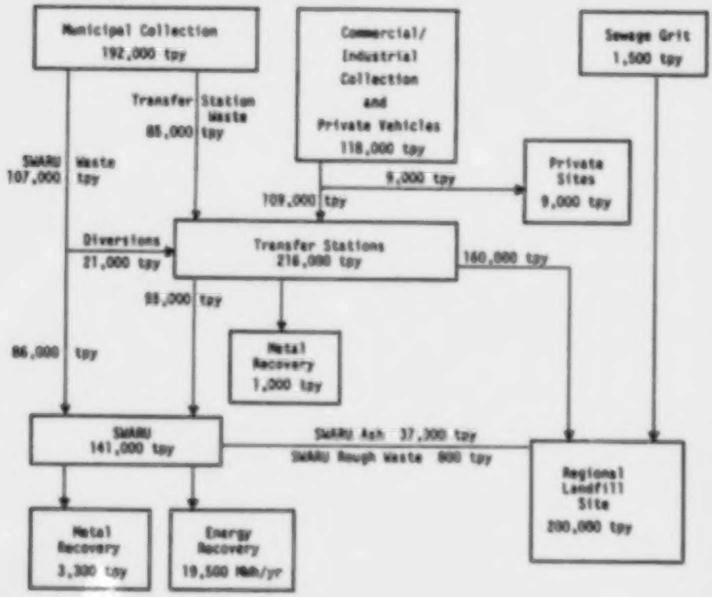
c) The Region's Waste Reduction Co-ordinator can liaise with private sector generators and haulers to arrange for separate collection of corrugated paper from sources that produce a significant quantity of this waste. Local haulers have expressed an interest in handling this material and local firms such as Domtar may provide readily accessible facilities for receiving depots or processing plants.

The recovered corrugated paper may be marketed through private sector firms, or processed through Third Sector Recycling, depending upon the specific strategy selected for transfer station separation.

The recovered wood may be directed to reduction for mulch, or shredded and processed for fuel at SWARU and other facilities.

FIGURE 1

Phase 1 Plus Phase 2A Waste Stream Flow:
Improved Waste Control
Plus
Transfer Station Floor Separation for SWARU


Municipal Collection
192,000 tpy
Commercial/
Industrial
Collection
and
Private Vehicles
118,000 tpy
Sewage Grit
1,500 tpy
Transfer Station
Waste
85,000 tpy
SWARU Waste
107,000 tpy
9,000 tpy
Private
Sites
9,000 tpy
109,000 tpy
Diversions
21,000 tpy
Transfer Stations
216,000 tpy
160,000 tpy
55,000 tpy
86,000 tpy
Metal
Recovery
1,000 tpy
SWARU
141,000 tpy
SWARU Ash 37,300 tpy
SWARU Rough Waste 800 tpy
Regional
Landfill
Site
200,000 tpy
Metal
Recovery
3,300 tpy
Energy
Recovery
19,500 MWh/yr

Analysis

Based upon data contained in the Region's 1987 "Solid Waste Management Overview" report, and supported by the field observations which were part of the SWARU Waste Routing Study, approximately 16,500 tonnes of corrugated paper are landfilled annually by the Region. Waste corrugated is generally not suitable for processing at SWARU, because the material usually arrives in an oversized and bulky condition. This characteristic would cause blockages in the SWARU conveying and shredding systems. Corrugated paper could be pre-shredded before introduction at SWARU, but it is considered to be a more valuable commodity for recycling rather than incineration. By co-ordinating efforts with either private sector haulers and recyclers, or the non-profit Third Sector Recycling agency which has established an extensive involvement in Regional househole recycling of paper, glass, and metal, corrugated paper recovery may be performed effectively.

With respect to wood reclamation, the measures outlined in Phase 1 for waste stream control are expected to reduce significantly the amount of wood entering the transfer stations. Because the actual impact in this regard cannot be determined adequately at this time, a separate wood bin can be installed when the Phase 1 measures are in effect. Records of the wood collected should be maintained for a period of one year, to establish a data base with respect to this waste stream. During that period, wood size reduction equipment may be used to process scrap wood, with the chips being used for local public works departments for mulch, at SWARU for fuel, or directed to private firms which manufacture commercial firelogs or other products using wood chips.

Corrugated Paper Recovery--

a) landfill requirements can be reduced by 8,000 tonnes annually or 3% (at 50% recovery) of the present landfill waste stream.

b) a useful and valuable commodity can be recycled, reducing the demand on virgin resources.

Wood Recovery--

a) landfill requirements can be reduced by about 8,000 tonnes per year, or 3% of the current landfill waste stream. This is an estimated amount, considered as the residual wood waste that will continue to arrive at the transfer stations after improved waste stream control eliminates the larger deliveries of wood debris, which will be diverted to private sector facilities. More accurate statistics will be available after a 12-month trail period, but a recovery rate of 50% of a typical 5% wood composition in waste is anticipated.

b) the recovered wood chips can be directed to useful applications for low-grade wood chip products, SWARU fuel, landscape mulch, or compost conditioner.

Figure 4 illustrates the waste stream flow that can be achieved by wood/corrugated reclamation.

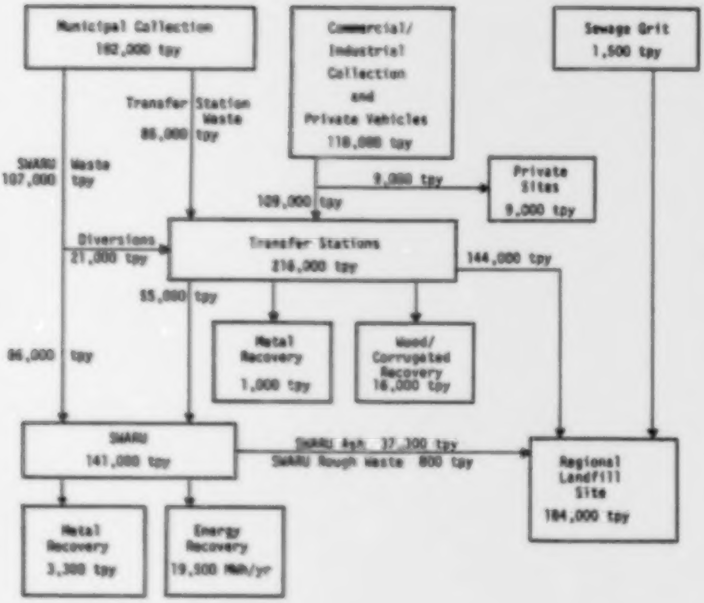
Costs

Corrugated Paper Recovery--

For a central depot, the cost of installing conveyors, screen, picking platforms, compactor and enclosure at the Renora Transfer Station

FIGURE 4

Phases 1, 2A and 2B Waste Stream Flow:
Improved Waste Control,
Transfer Station Floor Separation for SWARU,
Wood/Corrugated Reclamation


Municipal Collection
192,000 tpy
Commercial/
Industrial
Collection
and
Private Vehicles
118,000 tpy
Sewage Grit
1,500 tpy
Transfer Station
Waste
86,000 tpy
SWARU Waste
107,000 tpy
9,000 tpy
Private
Sites
9,000 tpy
109,000 tpy
Diversions
21,000 tpy
Transfer Stations
216,000 tpy
144,000 tpy
55,000 tpy
Metal
Recovery
1,000 tpy
Wood/
Corrugated
Recovery
16,000 tpy
86,000 tpy
SWARU
141,000 tpy
SWARU Ash 37,300 tpy
SWARU Rough Waste 800 tpy
Regional
Landfill
Site
184,000 tpy
Metal
Recovery
3,300 tpy
Energy
Recovery
19,300 MWh/yr

is estimated to be approximately $500,000. The incremental annual operating costs for labour, activities and maintenance is estimated to be about $150,000.

Wood Recovery--

The cost of installing minimal waste wood size reduction equipment such as grapple hog/chipper, conveyor and magnetic separator is estimated to be about $500,000.

Incremental annual operating costs, associated with the Kenora Transfer Station, are estimated to be approximately $100,000 for labour, utilities and maintenance.

FUEL PREPARATION PROCESS

Method

A fuel preparation process, illustrated in Figure 5, can be constructed at each transfer station, or at a central location, consisting of a receiving building large enough to accept one full day of waste deliveries, with storage capacity to accommodate an additional day's processed fuel product.

The key component of a simplified fuel preparation process or a refuse fuel upgrading system is a rotary drum screen (approximately 12.5 metres long by 3.6 meters diameter). The drum consists of 10-12 cm diameter holes.

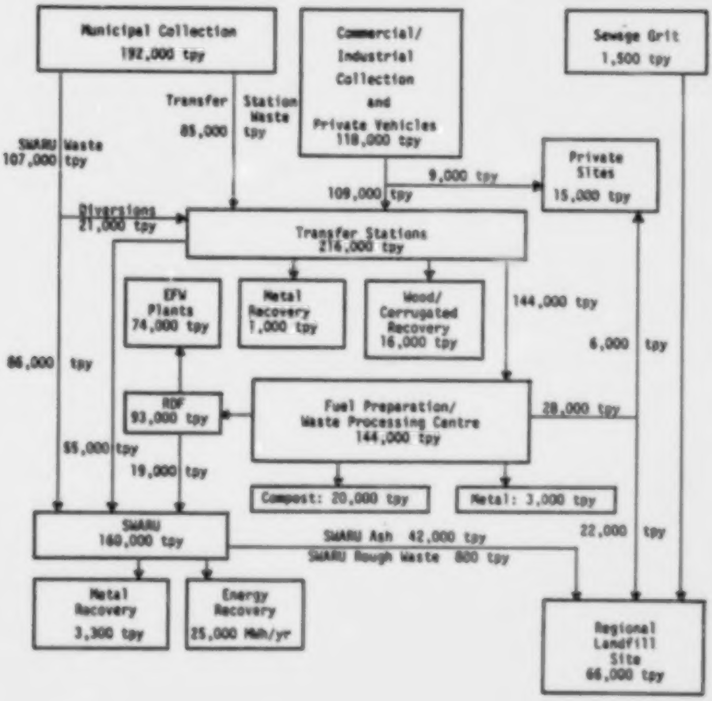
As the waste is turned in the drum, and it falls on the heavy duty plates and lifters inside the drum, the waste is liberated from its containers. The small material (unders) mostly the high ash, high moisture items falls through the holes and is separated. The larger material (overs) will consist of relatively low ash, low moisture items having a BTU value of greater than 6,500 BTU/lb compared to as received waste which is about 4,800 BTU/lb. The anticipated mass balance for the drum screen for mixed municipal solid waste is as follows:

TABLE 2 % Composition

Component	Waste As Received	Unders	Overs
paper	45	5	40
plastic	6	1	5
food waste	12	8	4
yard waste	12	7	5
wood	6	1	5
other combustibles	4	2	2
metals	5	3	2
glass	7	6	1
other non-combustibles	3	2	1
TOTAL	100	35	65

FIGURE 5

Phases 1, 2A, 2B and 3 Waste Stream Flow:
Improved Waste Control,
Transfer Station Floor Separation for SWARU,
Wood/Corrugated Reclamation,
Fuel Preparation/Waste Processing Center


Municipal Collection 192,000 tpy
Commercial/ Industrial Collection and Private Vehicles 118,000 tpy
Sewage Grit 1,500 tpy
Transfer Station Waste 85,000 tpy
SWARU Waste 107,000 tpy
9,000 tpy
109,000 tpy
Private Sites 15,000 tpy
Diversions 21,000 tpy
Transfer Stations 216,000 tpy
EFW Plants 74,000 tpy
Metal Recovery 1,000 tpy
Wood/ Corrugated Recovery 16,000 tpy
144,000 tpy
6,000 tpy
86,000 tpy
RDF 93,000 tpy
Fuel Preparation/ Waste Processing Centre 144,000 tpy
28,000 tpy
55,000 tpy
19,000 tpy
Compost: 20,000 tpy
Metal: 3,000 tpy
SWARU 160,000 tpy
SWARU Ash 42,000 tpy
SWARU Rough Waste 800 tpy
22,000 tpy
Metal Recovery 3,300 tpy
Energy Recovery 25,000 MWh/yr
Regional Landfill Site 66,000 tpy

The unders can be further processed (magnetic separation, screening, size reduction, digestion) to recover metals and compost.

The overs can also be further processed (picking, coarse size reduction and magnetic separation) to recover paper, plastics, metals and improved refuse fuel.

Conceptually, drum screens could be installed at the transfer stations. A central waste processing facility is deemed more practical especially if it is located adjacent to SWARU. The processing facility could then be considered an extension of SWARU and could provide the needed storage capacity for SWARU.

Analysis

In accordance with the findings of the SWARU Waste Routing Study, it has been determined that recommendations proposed in the Phase 1, Phase 2A, and Phase 2B strategies are more than adequate to supply all the acceptable waste necessary to maximize use of SWARU and minimize landfill requirements. In fact, these measures are expected to provide a surplus of SWARU quality fuel that will sometimes exceed the process capacity of the plant. Also, these strategies can be effected directly and easily as they require little lead time, involve basic technology and administrative measures, and are relatively low-cost.

For these reasons, it was recommended that the Region pursue actively the immediate implementation of the three noted strategies. However, to maintain its progressive leadership in the field of waste management, the Region should consider a fuel preparation/upgrading facility in the near future to further reduce its dependence on landfill. In conjunction with the three strategies for current application, and the Blue Box program, a complete refuse processing facility would reduce landfill requirements by approximately 75% of current demand.

Because of the extensive approvals, detailed design, construction, commissioning, financial, and market needs associated with a central waste processing plant, a lead time of five year for planning and design is recommended. This project would also lend itself readily to development in parallel with a refuse/sludge composting program, consistent with the Region's current directions in this regard.

The issue concerning final disposition of the enriched fuel stream should be examined briefly at this time. It is estimated that the refuse stream available for the waste processing plant, remaining from the implementation of Phases 1, 2A, and 2B, is approximately 141,000 tonnes per year. An estimated upgraded fuel stream of approximately 93,000 tonnes could subsequently be reclaimed from this balance. The other 48,000 tonnes would be divided among material recovery, compost, and landfill fractions. Of this, about 28,000 tonnes constitutes the landfill stream. It may be possible to further reduce the 28,000 tonnes per year landfill demand by 20% or more by transporting classified components of the landfill fraction to certified local private facilities for quarry rehabilitation.

Alternately, the total municipal collection of 192,000 tonnes may be directed to the refuse processing facility. This measure can provide significant benefits in terms of improved SWARU waste quality, greater energy recovery, and reduced ash residue. However, the capital and operating costs for a processing plant sized to handle the total waste stream will be greater than the costs of processing the balance of wastes remaining from the implementation of Phases 1 through 2B. It is recommended that this alternative be evaluated in detail when the fuel preparation process concept is developed.

With the available high-quality surplus fuel stream, the Region may consider expanding capacity at SWARU, installing one or more smaller current energy-from-waste facilities at the new refuse processing plant site, or marketing the fuel product (RDF) to commercial and industrial users. There are already a number of pilot projects employing RDF established within an economical haulage distance, and these users constitute an attractive option for the Region's long-range waste management planning.

Benefits

The operation of a central waste processing plant would provide an annual additional recovery of 93,000 tonnes/year of high-grade RDF, reclaim approximately 3,000 tonnes of metal and provide about 20,000 tonnes of compost feedstock. In addition, the classified separation of inert wastes that can be directed to private quarry disposal sites will reduce annual landfill requirements to about 66,000 tonnes, based upon the present waste stream.

Costs

The capital cost of a refuse processing/fuel upgrading and storage facility (500 tonnes/day) is approximately $10 million with an annual operating expense of about $2 million.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

RESOURCE RECOVERY IN PALM BEACH

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ABSTRACT

Palm Beach County's resource recovery facility is described, outlining the planning process and technology selection. Major systems are described with emphasis on refuse processing, materials recovery, fuel feed, boiler technology, air quality control, and steam-electric cycle.

In January 1987, the Solid Waste Authority of Palm Beach County contracted with Palm Beach Energy Associates, a joint venture of The Babcock & Wilcox Company and Bechtel Civil, Inc. to design, procure, and construct its 2000 tons/day resource recovery facility. The facility is currently in startup. Babcock & Wilcox alone will operate the facility under a long-term operating contract. National Ecology, Inc., under separate subcontracts to Babcock & Wilcox, designed, furnished, and will operate the refuse-derived fuel (RDF) processing system.

PLANNING AND IMPLEMENTATION

BACKGROUND

Palm Beach County is located on the southeast coast of Florida and is the state's largest county, occupying an area of approximately 2,000 square miles and extending approximately 45 miles along its north-south axis and 53 miles along its east-west axis (Figure 1).

In respect to its resident population, Palm Beach County has exhibited continuous and accelerated growth. As reported in the 1910 Census, the county maintained a population of 5,577 persons and ranked 39th among Florida counties. By 1920, Palm Beach County had grown to rank 17th and by 1930, with a population of 51,781, it had grown to be the seventh most populous county in the state. Now, it is Florida's fifth most populous county, with a population of approximately 770,000, and has one of the most rapid growth rates in the nation (Figure 2).

Despite its relatively vast area, Palm Beach County does not possess abundant land resources for its citizens because of its rather unique population distribution (Figure 1). The population remains concentrated in two relatively small areas; one is a narrow band along the county's eastern coastline, the other a similarly narrow band along the shore of Lake Okeechobee in the county's northwestern corner. More than 90 percent of the county's population lives in the eastern coastal strip.

Consequently, the county has been pressed to meet the demands of continued development and rapid population growth in the supply of necessary services. Developing appropriate long-range solid waste management plans and providing continuous high quality solid waste management services has proven to be a substantial challenge.

FORMATION OF THE SOLID WASTE AUTHORITY

Prior to the last 10 to 15 years, solid waste management in Palm Beach County had been undertaken in what would have to be characterized as a haphazard fashion. For example, in 1966, there were 13 authorized landfills and numerous illegal dumps being operated within the county. Four years later, the Palm Beach County Area Planning Board reported that some 350 unauthorized dumps were identified throughout the county. Clearly, the problem of effectively managing the county's solid wastes was getting out of hand.

In 1975, to deal with the mounting problem posed by waste management needs, the State of Florida's Legislature created the Solid Waste Authority of Palm Beach County. Since that time, the Authority has been responsible for the development of plans and programs to provide for effective countywide solid waste management. Initially, the Authority's role was planning and administrative only. In October 1983, the Authority moved into an operational role.

Specifically, the Authority is responsible to:

1. Provide for the safe and sanitary processing and disposal of solid waste.
2. Provide a coordinated countywide program for the management of hazardous waste and control of solid waste processing and disposal in cooperation with federal, state and local agencies responsible for the prevention, control, or abatement of air, water and land pollution.
3. Require the municipalities and the county to plan for and develop an adequate solid waste collection system.

LONG-RANGE PLAN

While the process of developing a long-range solid waste management plan for Palm Beach County can be traced back as far as 1971, activities began to take on a new dimension in 1977. At that time, the Authority began to study a regional approach to solid waste management, integrating recycling, resource and energy recovery, and transfer station and landfill facilities.

Initially, the Authority evaluated the condition of the solid waste management system serving the county's sparsely populated Glades Region, followed by an evaluation of the rapidly growing eastern portion of the county. These studies resulted in development of the 1979 Palm Beach County Comprehensive Solid Waste Management Plan, entailing a multi-phased approach of: a) data collection, b) data analysis, c) systems analysis, d) development of alternative plans, and e) recommended plan.

During the development of alternatives, three plans for the development of controlled landfills offered potential for both short- and long-term benefits to the county. In addition, four reliable resource recovery systems were identified for consideration. These systems were based upon the thermal conversion of solid waste to steam and electricity and the production of refuse-derived fuel for export to a local utility's power plant.

In 1979, when the report was completed, the abundance of low cost natural gas coupled with the optimism regarding coal availability presented significant obstacles to the development of a cost-effective resource recovery project in Palm Beach County. As a result, it was determined that the high tipping fees necessary to offset a low income stream from a resource recovery project resulted in its economic infeasibility. A system which relied exclusively upon sanitary landfills was therefore recommended. However, the plan did recommend that resource recovery options be re-evaluated in 1982.

In 1980, the Authority initiated studies to identify suitable landfill sites and review a variety of available resource recovery options. In 1981, the Authority instituted a System Integration Planning Process to determine the technical, economic, institutional and environmental considerations associated with implementation of its Comprehensive Solid Waste Management Plan.

The results of the studies and planning process were compiled into a Feasibility Study Report which was completed in 1982. As an outcome, it was determined that, due to the rapid rise in the costs of energy and landfilling, the development of a resource recovery project would represent a cost-effective solid waste management alternative.

The 1982 report provided the Authority with the basis for a decision on the direction for the development of its long-term solid waste management system. Evaluations were made of facility siting alternatives, technology options, waste characterization, market identification, and alternative system configurations. Inherent to the Authority's decision to proceed with the rehabilitation of its waste management system was an evaluation of state-of-the-art waste management technologies, including:

- Landfilling
- Waste-to-Energy

    - Modular Incineration
    - Mass Burn
    - Refuse-Derived Fuel
    - Pyrolysis

- Composting

As a result of this evaluation, waste-to-energy, utilizing refuse-derived fuel (RDF) technology, was chosen to be employed in conjunction with a continuation of landfilling to best meet the long-term needs of the county. The Authority's decision to proceed with RDF technology was based upon its belief that such technology provided a flexibility not found in the other systems. Additionally, the Authority believed that the application of RDF technology offered environmental benefits above and beyond those provided through the other technologies considered.

The Authority, having concluded that resource recovery was indeed feasible and desirable for application to the county's waste management needs, proceeded with finalizing site selection and resource-recovery-technology selection. An implementation schedule was formally adopted, influenced by the specter of an ever-growing waste stream.

SITE SELECTION

The Authority's most difficult and time-consuming task (approximately 7 years) was that of site selection. In addition to the obvious environmental concerns which had to be addressed, the site's location had to represent a balance between economic and operational constraints, and real and perceived public impact. Some 30 to 40 sites were evaluated throughout the county and all but a few of these sites were eliminated.

After the technical and political feasibilities were analyzed, a viable site was selected for the project. Then, the Board of County Commissioners, in attempting to strike an acceptable balance, adopted a resolution to relocate the site approximately two miles north to an alternate site, the location of the current project. The Commissioners also agreed to transfer all solid waste facilities currently operated by the county to the Authority, which the Authority accepted.

FINANCING

Once plans had been developed, the pressing issue of system financing came to the forefront. Phase I of the project consisted of improvements to existing transfer stations and landfill facilities and the construction of new transfer station and landfill facilities. Additionally, the site for a resource recovery project was to be acquired. The projected cost of this phase of the project was set at $43,500,000. The Authority prepared a contract between itself and the Board of County Commissioners and a bond resolution for Phase I Project Implementation. Shortly thereafter, technical consultants to the Authority prepared an Engineering Feasibility Study for the Phase I Bond issue. Phase II and Phase III Bond issues followed in 1985 and 1986, in amounts of $320,000,000 and $100,000,000, respectively, to fund the remainder of the Authority's programs and retire the Phase I Bonds.

CONTRACTOR SELECTION

In 1985, the Authority undertook and completed a comprehensive study of the costs and benefits associated with private versus public ownership of a waste-to-energy facility. Consequently, the Authority determined that public ownership was the preferred course of action for Palm Beach County.

Subsequent to its decision to undertake a public procurement, the Authority prepared and published a Request for Qualifications (RFQ) for the design, construction and operation of its North County Regional Resource Recovery Facility. After a complex series of reviews of responses to the RFQ, involving a citizens committee, Authority staff and consultants, and third-party experts, a list of "qualified vendors" was prepared.

Later in 1985, the Authority prepared and issued a detailed Request for Proposals (RFP) for the design, construction and operation of its planned waste-to-energy facility to the four vendors which had qualified.

Throughout 1986, the Authority evaluated the responses received from the four vendors to the RFP, culminating in a decision to enter into a simultaneous contract negotiation process to determine the firm most responsive to the Authority's needs. In the end, the partnership of Babcock & Wilcox and Bechtel (Palm Beach Energy Associates) was selected as the primary vendor for detailed contract negotiations. On December 11, 1986, the Authority entered into a Design and Construction Agreement with Palm Beach Energy Associates for the North County Regional Resource Recovery Facility for a contract construction price of $171,351,000. On that same date, an Operations and Maintenance Agreement was entered into with Babcock & Wilcox.

MANAGING THE PROJECT

The project team is composed of the following major participants:

Solid Waste Authority of Palm Beach County and its consultants are responsible for site support facilities such as potable and non-potable water wells, wastewater injection systems, external roads, land management planning and contractual coordination of the electric power sales agreement.

Palm Beach Energy Associates (PBEA) has overall responsibility to the Authority for design, procurement, and construction of the resource recovery facility. PBEA is managed by a four-member Board of Control, Project Manager and Deputy Project Manager. The workscope responsibility of PBEA is generally divided between Babcock & Wilcox and Bechtel Civil, Inc. under a partnership agreement as follows:

Babcock & Wilcox (B&W) is responsible for design, procurement, and construction of the refuse-derived fuel plant and boiler plant. B&W is also responsible for overall facility startup, testing, and precommercial operation. The design, procurement, and precommercial operation of the refuse-derived fuel processing system are subcontracted to National Ecology, Inc.

Bechtel Civil, Inc. is responsible for design, procurement, and construction of the electric generating plant, balance-of-plant systems, site work, civil/structural work, architectural, and electrical distribution.

B&W alone is responsible to the Authority for the long-term operation and maintenance of the resource recovery facility. B&W has subcontracted the operation and maintenance of the RDF processing system to National Ecology.

These diverse organizations were located across the USA. A project management team was established at the jobsite in Palm Beach County within three months of Notice to Proceed. This helped to integrate and coordinate the complex relationships and systems that crossed the boundaries of several participants. Monthly schedule meetings between the principal firms were also held to facilitate engineering coordination and construction activities.

SCHEDULE

The Authority initiated Notice to Proceed on January 7, 1987. The Design and Construction Agreement specifies a 36-month construction period. Based on present estimates, the project may be completed early. Project milestones are:

First Concrete	June, 1987 (actual)
Lift Boiler Drums	November, 1987 (actual)
Set Main Transformer	May, 1988 (actual)
Tie-in with Electric Utility	September, 1988 (actual)
Energize 480 V Switchgear	October, 1988 (actual)
First Boiler Fire	April, 1989 (planned)
First RDF Fire	May, 1989 (planned)
Turbine Roll	May, 1989 (planned)
Acceptance	Late Fall, 1989 (planned)

FACILITY DESCRIPTION

The Palm Beach County resource recovery project consists of three major subprojects: the resource recovery facility, landfills, and support service facilities (Figure 3). The resource recovery facility itself consists of three major plants: the refuse-derived fuel plant, the boiler plant, and the electric generating plant.

The facility will process 2,000 tons/day of municipal solid waste (MSW) with an annual throughput of 624,000 tons. Excess capacity and redundancy were built into the facility. The facility is equipped with three RDF lines, any two of which can handle 2,000 tons/day of MSW. Two boilers are provided, each with a capacity to burn 900 tons/day of RDF, which is the equivalent of 1085 tons/day of MSW. The turbine-generator capacity is matched to the full output of the boilers. In addition, a steam dump condenser is provided to permit full load boiler operation in the event of a turbine-generator outage. Separate 3,000-ton storage areas are provided for MSW and RDF to decouple and levelize the operation of the boiler and electric generating plants from variations in daily MSW delivery and the operation of the RDF plant. Two control areas are provided, one for the RDF plant and one for the boiler and electric generating plants. The Authority envisions expansion of this facility to 3,000 tons/day capacity, accommodated by the addition of a third boiler and second turbine-generator.

A schematic flow diagram of the facility is presented in Figure 4 and the facility layout is shown in Figure 5.

The facility can handle a wide range of MSW composition and heating value. The Reference Waste used as the design basis is shown in Table 1.

The MSW is delivered to the facility by trailers from transfer stations and packer trucks. It is then processed into refuse-derived fuel in the RDF plant. Separation of non-combustibles, including glass, sand, grit, rock, and metals, takes place while maximizing the recovery of the MSW's fuel value. For each 100 tons of MSW processed, approximately 83 tons of RDF result. Fuel value recovery is approximately 96 percent; that is, 96 percent of the Btu's in the incoming MSW are in the product RDF. Ferrous metal and aluminum can recovery are 90 percent and 60 percent, respectively. From 100 tons of MSW, approximately five tons of ferrous metal and one ton of aluminum cans are recovered. Processing MSW results in refuse-derived fuel, the analysis of which is shown in Table 1.

The RDF is burned in the two boilers, generating superheated steam at 750 psig, 750°F. At full load, each boiler produces 310,000 lb/hr of steam which is sent to a 61,300 kW turbine-generator. The power cycle consists of a water-cooled condenser with cooling tower and three feedwater heaters.

The net electric output of the facility, when operating at the nominal 2000 tons/day rating, is 563 kWh/ton of Reference MSW. The electricity is sold to the local utility under a long-term contract.

RDF TECHNOLOGY, THE THIRD GENERATION

The first generation of RDF plants, built in the early 1970's, was based on the "crunch and burn" concept in which municipal solid waste is shredded, ferrous metal removed with magnets, and the balance burned in a dedicated boiler. This technology, which is still in use today in some facilities, is relatively simple and provides almost 100 percent recovery of combustible materials. However, the RDF has a very high ash content and also includes all of the glass, stones, grit, dirt, and aluminum and other non-ferrous metals in the original incoming waste. This affects boiler maintenance costs, as well as loss of revenue from the sale of recovered aluminum. Second generation RDF systems incorporated screens and air classifiers to reduce the ash content of the RDF, resulting in a higher Btu, lower ash RDF, but with a lower mass yield and lower total fuel value recovery. The Palm Beach County facility is a third generation "state-of-the-art" plant that will recover 96 percent of the combustibles from the incoming waste. It will also maximize the recovery of ferrous metal and aluminum cans for sale. The engineering design might be described as "win-win" in that it provides for a very high yield of relatively low ash RDF, together with efficient materials recovery.

RDF PLANT DESIGN BASIS

The design is based upon processing 2,000 tons of MSW daily during a 16-hour period, with an additional eight-hour shift dedicated to maintenance activities. The facility incorporates three identical processing lines, two of which will normally operate with the third redundant line available if one of the other two lines is out of service. In addition, an oversized bulky waste (OBW) line and a tire shredding line are provided. The design of the facility is rugged and simple, and based on technology and components with a proven record of performance.

A mass flow diagram showing movement of the waste through the RDF plant is provided in Figure 6. This diagram is based on processing 12,000 tons/week of MSW over a period of 11 shifts, at an average throughput rate of 76 tons/hour. The tipping floor will hold approximately 3,000 tons of MSW, which is 1,000 tons more than the expected daily throughput. During typical operating conditions, the MSW storage area will be emptied daily. A separate building capable of storing 3,000 tons of RDF is also provided.

A schematic drawing of one of the three RDF processing lines, the oversized bulky waste line and the tire shredder line is provided in Figure 7.

INFEED CONVEYORS

From the tipping floor, the solid waste is fed by front-end loaders to steel-pan apron conveyors. These conveyors feed the flail mill infeed conveyors from which unacceptable or hazardous waste is removed by means of a grapple crane. The operator of the grapple crane can vary the speed of these conveyors to meter MSW to the flail mill at a uniform rate.

INITIAL SIZE REDUCTION

The 400 hp, 100 ton/hour flail mill tears open the plastic garbage bags, shreds the refuse, and also breaks glass bottles to a size of approximately 2 inches or less. In this manner, the flail mill reduces larger items in the refuse stream to an easily conveyed size, while avoiding an undesirably fine grind at this stage of the process.

FERROUS METAL RECOVERY

Ferrous metal is then extracted from the coarsely shredded MSW in each line by a single-stage overhead magnet. Each magnet system is enclosed for noise and dust control. Recovered ferrous metal is conveyed to a ferrous storage area from where it will be further processed through the OSW mill to remove tramp materials such as paper, plastics or textiles, thereby providing a cleaned ferrous metal.

SIZE CLASSIFICATION AND FINAL SIZE REDUCTION

After ferrous removal, shredded waste is fed into a rotating trommel screen 10 feet in diameter by 60 feet long. The trommel performs the following functions:

- Removes glass, sand, grit, dirt and non-ferrous metal less than 2 inches in size.
- Removes the minus 6 inch fraction which contains the bulk of the aluminum cans.
- Produces a high fuel value, low ash oversize fraction.

The dimensions of the openings in the screen can be varied if necessary to fine-tune the processing system thus assuring maximum recovery of combustibles consistent with low ash content. The trommel is fully enclosed for noise and dust control, and is also equipped with a water sprinkler system for fire protection.

The boilers require a nominal 4-inch-size RDF fraction, part of which is provided directly by the trommel undersize, and part by shredding the trommel oversize in a horizontal secondary shredder, equipped with a 1,000 hp, reversible, variable speed motor. Since the secondary shredder is a major consumer of energy and has high hammer maintenance costs, the RDF process is specifically designed to reduce the shredder load by secondary shredding only those combustibles too large for the boiler. RDF particle size is controlled with a disc screen which recycles oversize material to the secondary shredder.

SEPARATION OF GLASS, STONES, GRIT, AND DIRT

Trommel undersize from the first section of the trommel is essentially minus 2 inches in size. This undersize material passes over an air density separator (ADS) designed to remove dense particles from less dense materials through a combination of vibration and air sweeping. This device can efficiently remove glass, stones, grit and dirt, as well as other non-ferrous metals from the combustibles. The ADS can be adjusted to vary the amount of heavy fraction. This is desirable to compensate for variations in the moisture and inerts (ash) content of the incoming municipal waste.

The light fraction, which can range from approximately 50 percent to 90 percent of the ADS feed, consists essentially of combustibles with fuel value ranging from 3,500 Btu/lb to 5,000 Btu/lb depending on the composition of incoming waste and ADS settings. This light fraction is blended into the RDF.

ALUMINUM CAN RECOVERY

To optimize conditions for aluminum can recovery, an air classifier is provided for the plus 2 inch, minus 6 inch trommel undersize fraction. The air classifier will remove the light organic portion of the stream thus allowing aluminum cans to be more visible for removal by hand pickers.

The air classifier heavy fraction drops onto a 48-inch-wide conveyor moving at approximately 30 feet/minute with eight hand-picking stations on either side of the belt. Hoppers are provided at each of the picking sites; cans deposited in them drop to a conveyor feeding the can flattener. A pneumatic conveyor then transfers the cans into a trailer. Space has been provided for the possible future addition of an eddy-current separator for the removal of aluminum cans.

OVERSIZED BULKY WASTE AND TIRE SHREDDING LINES

The OBW shredder is a 1,000 hp horizontal hammer mill that will be used to shred ferrous metal recovered by the RDF processing lines, white goods (such as refrigerators and washing machines) and OBW (such as furniture and tree limbs). Ferrous metal shredded by the OBW shredder will be separated from extraneous materials and non-ferrous metals by an overhead single-stage magnet. The ferrous metal will then be given a final cleaning to remove tramp materials by the use of an air scrubber.

A shear mill designed specifically for shredding tires is provided with a capability of shredding 500 passenger tires per hour. The mill includes a rotary screen classifier for recycling shredded tires that exceed 2 inches, thereby delivering a material 2 in. x 2 in. or less to the RDF conveying system.

DUST CONTROL

Each processing line, along with the OBW line, has an independent dust control system. All conveyor transfer points and dusty equipment are enclosed, sealed with rubber flaps, and aspirated to collect fugitive dust. Each processing line has a separate dust control system including a baghouse. In addition, a dust control system with a baghouse is provided to collect fugitive dust and debris from the ferrous metal air scrubber. The collected materials are added to the RDF.

CONTROL SYSTEM

The three processing lines along with the OBW line and tire line will each have independent controls. During process operation, each piece of equipment will be sequentially controlled by a programmable logic controller. In the event that a piece of equipment malfunctions, all components upstream of the malfunction (with the exception of the shredder) will shut down immediately. Once the problem is rectified, the system will be restarted in sequence beginning at the outage proceeding to the front end. By way of minimizing the potential for additional jams caused by product surges, each piece of equipment has a controlled start mode such that it will not operate until all downstream components are running.

BOILER TECHNOLOGY - THE THIRD GENERATION

Early experience burning refuse was limited to refractory incinerators followed by low pressure waste heat boilers. Babcock & Wilcox supplied such a waste heat boiler to the City of Miami, Florida in 1930 and similar boilers to many other municipalities. Refuse boilers have since evolved into today's modern waterwall boilers with integral combustion chambers operating at the high steam pressures and temperatures needed for efficient electric power generation.

The drive to develop RDF boiler technology had the goals of providing a more efficient, more load responsive, and more economical facility. RDF has been burned since the late 1960's as a supplemental fuel in boilers and in dedicated RDF boilers. Many lessons were learned from the first and second generation plants. Babcock & Wilcox RDF boiler experience represents over 170 boiler-years of operating experience or 55 percent of the total number of all RDF boilers. The learning process typical of new technology has taken place to enable the third generation of RDF boilers for the Palm Beach County facility and others like it to be built (Figure 8).

COMBUSTION TECHNOLOGY

The composition of municipal refuse is very similar to that of wood on an ash-free basis. However, the ash content of wood is very low compared to MSW. With the removal of ferrous metal and other non-combustibles, RDF can be burned in a boiler of the same basic design as that used for waste wood.

The RDF is fed onto the boiler's traveling grate stoker. In addition to providing a platform on which the refuse can burn and convey away the ash, the grate serves as a means to introduce part of the combustion air to the furnace. This undergrate air helps dry the fuel and agitate the fuel bed. The balance of the combustion air is injected through overfire air ports above the grate. The placement and velocity of the overfire air ports is critical to achieving complete combustion.

The efficient combustion of any fuel requires the three "T's" of combustion - time, temperature, and turbulence. To achieve the three "T's" in a stoker fired boiler requires careful coordination between stoker and furnace design parameters. What was missing from the early RDF units was optimum turbulence.

In the early 1970's, Babcock & Wilcox developed the Controlled Combustion Zone (CCZ) furnace for the combustion of high moisture wood. A diagram of the CCZ furnace design is shown in Figure 9. B&W's experience base showed that furnace arches to clearly define the combustion zone should be used for hard-to-burn fuels.

Twelve CCZ boilers are currently in operation. Prior to the CCZ design, wood and RDF fired boilers typically utilized ash reinjection systems to reduce unburned carbon content of the ash to acceptable levels. Reinjection systems were effective in maintaining low carbon loss; however, the ash load going through the boiler increased erosion considerably. The success of the CCZ design is demonstrated by the fact that no reinjection system is required to maintain low unburned carbon loss.

FUEL FEED SYSTEM

Uniform fuel feed into the boiler is essential to maintain stable boiler load and an adequate cover of ash on the stoker grate. This ash bed is needed for two reasons: 1) it protects the grate from overheating, and 2) any aluminum remaining in the RDF, which melts in the high temperatures of the furnace, solidifies in the ash bed before reaching the grate and causing jamming of the moving grate bars and plugging of the air holes.

The operation of many early fuel feed systems was quite troublesome, resulting in fuel feed interruptions. Several first generation facilities required wholesale modifications to their fuel feed systems, evolving into the proven system to be used at the Palm Beach County facility.

Redundant conveyors from the RDF plant to the boilers are provided. RDF may be taken directly from the RDF processing lines or reclaimed from the RDF storage area. This system utilizes a "run-around" design, where more fuel is conveyed to the boilers then is actually dropped into the six individual ram-type fuel feeders on each boiler. The excess fuel is returned to the RDF storage building by another set of conveyors. In this manner, the boiler fuel demand is always fully met.

CORROSION PROTECTION

Corrosion was not a problem in first generation boilers which typically produced low pressure saturated steam. With the second generation of RDF boilers, steam pressures and temperatures rose to meet the requirements of electric power generation cycle conditions, typically 600+ psig, 750+°F. Early operating experience of second generation boilers saw elevated corrosion rates; forced outages sometimes occurred within months of startup.

Corrosion in refuse boilers is related to the chlorides, metals, and ash in the fuel. Boiler tube metal temperature is slightly higher than the water or steam within the tube. The rate of tube metal corrosion is a function of metal temperature, the corrosion rate rapidly increasing with higher metal temperatures resulting from operation at steam pressures necessary for efficient electric power production.

Another cause of corrosion is that localized areas of the lower furnace alternate between an oxidizing and reducing atmosphere. This is caused by imperfect mixing of the fuel with the air. The CCZ boiler design provides for better mixing of fuel and air, thereby minimizing the oxidizing/reducing corrosion potential.

To solve corrosion problems in the lower furnace, refractory coatings were tried. Unfortunately, the refractory caused higher furnace temperatures which resulted in slagging problems. High alloy nickel materials have also been used. Weld overlay is one way to apply the high alloy material; this is particularly useful in a retrofit situation. The Palm Beach County boilers have special bimetallic furnace tubes, nickel alloy for corrosion protection on the outside, and carbon steel on the inside for strength.

AIR QUALITY CONTROL SYSTEM

The air quality control system utilizes dry scrubber/electrostatic precipitator technology to control acid gases and particulate. The air permit requirements include limitations for SO_2 of 65 percent removal or 0.32 lb/MMBtu; for HCl, 90 percent removal; and for HF, 0.0032 lb/MMBtu. The outlet particulate limit is 0.015 grains/dscf corrected to 12 percent CO_2 and the outlet opacity limit is 15 percent.

The system consists of a spray dryer absorber followed by a high efficiency electrostatic precipitator for each boiler (Figures 4 and 5). Each precipitator has four fields; three fields are normally in operation with the fourth as a spare.

Flue gas laden with fly ash leaves the boiler at 350°F and enters the top of the spray dryer absorber. Within the spray dryer, flue gas is brought into turbulent contact with a fine lime-water slurry spray. As the water in the lime slurry evaporates, the flue gas is cooled to about 250°F. Calcium hydroxide ($CaOH_2$) in the lime slurry reacts with the HCl and SO_2 in the flue gas to form primarily calcium chloride ($CaCl_2$) and calcium sulfite ($CaSO_3$). A portion of the fly ash, excess lime and reaction-products mixture drops out in the spray dryer hoppers, but the major portion is carried into the electrostatic precipitator. The cleaned flue gas is then drawn through the induced draft fan and exhausted to the 250-foot stack.

The reagent preparation system consists of two 100-percent-capacity lime slaking systems for each boiler train. Pebble lime is delivered by trucks and pneumatically conveyed into the lime silos. Lime and water are added to the slaker in proportions to maintain slaking reaction temperatures. The slurry from the slaker passes through a grit screen before entering the slurry storage tank. The lime slurry is pumped from the slurry storage tank to the spray dryer at the rate required to maintain stack effluent requirements. Additional water is added to the lime slurry upstream of the atomizer to control the outlet gas temperature of the spray dryer. .

ASH HANDLING SYSTEM

The reliability and redundancy criteria used in other materials handling systems in the facility were also used in the design of the ash handling system.

Bottom ash from each boiler is discharged through a diverter valve to one of two 100-percent-capacity water bath deashers. The bottom ash is dewatered as it moves up the inclined end of the deasher and discharges onto the bottom ash transfer conveyor, which moves the material to the ash building, where the bottom ash is mixed with fly ash at truck loadout.

The fly ash and spent lime are discharged from the spray dryer and precipitator hoppers through rotary air locks into dry drag conveyors. These drag conveyors move the material to either of two 100-percent-capacity transfer drag conveyors for transport to the ash silo. The fly ash from the silo is moistened in a pug-mill fly ash conditioner for dust control just prior to the bottom ash mix point and then is discharged into trucks.

The bottom ash/fly ash mixture is hauled to the adjacent lined landfill. Ash utilization presently is being studied by the Authority.

ELECTRIC GENERATING PLANT

CYCLE DESCRIPTION

The electric power plant incorporates a conventional Rankine steam turbine cycle illustrated in Figure 10. High pressure, superheated steam from the boilers is expanded through a proven, 14-stage turbine design. Turbine exhaust steam is condensed at 2-3 in. Hg abs. pressure, depending on ambient temperatures, in the water-cooled condenser. Two 100-percent-capacity condensate pumps draw water from the condenser hotwell and pump the condensate through the low pressure feedwater heater and then to a tray-type deaerating heater. Three 50-percent-capacity boiler feed pumps are fed from the deaerator storage tank; two feed pumps are motor driven, one is steam turbine driven. The feed pumps supply water through a high pressure feedwater heater to the boilers.

The electric generator is an air- and water-cooled synchronous machine, driven at 3,600 rpm by the steam turbine. It develops a maximum of 61,300 kilowatts at 13,800 volts. Generator electric output is stepped-up to 138,000 volts through the main station transformer which supplies the local utility's 68 MVA switchyard.

The full-flow steam dump condenser is a unique feature of the steam cycle. This U-tubed heat exchanger allows the RDF plant and boilers to perform independently of the turbine-generator and auxiliaries, thereby increasing availability of the plant. The dump condenser is furnished with corrosion-resistant titanium materials.

WATER SYSTEMS

Cooling water for the condenser is supplied from a three-cell, counterflow cooling tower by three 50-percent-capacity circulating water pumps. Cooling water for plant auxiliaries is also drawn from the discharge of the circulating water pumps. Makeup for the cooling tower is provided from interceptor wells that pump water from the adjacent landfill's ground water withdrawal system. Cooling tower blowdown water is discharged to a deep well injection system, which pumps the water into 3000-foot-deep wells in the substratum rock beneath the plant.

Water for fire protection is self-contained within the facility. A 500,000 gallon fire water tank stores water that is piped throughout the plant through two 100-percent fire pumps, one motor-driven and one diesel engine driven.

Potable water for process and personnel use is provided by a local city water supply system and stored in above-ground tanks.

The demineralized water system for boiler makeup uses potable water as influent, with treatment through organic filters and strong anion and cation vessels. Wastewater from the demineralized water system is stored, treated and pumped to the deep well injection system.

The plant storm water runoff system was designed in close coordination with the South Florida Water Management District. Runoff from the site is controlled through the use of water retention areas and discharge control structures.

BALANCE OF PLANT

Electric auxiliaries are furnished through the plant electrical distribution system. Electric power from the main generator is stepped-down to 4160, 480 and 120 volts for powering auxiliary loads. A 125-volt dc system is provided for control purposes and emergency services, including a 120-volt uninterruptible power supply. A 200 kW emergency diesel generator is also provided for use during shutdown service. In-plant communication is accommodated with a multi-party paging system.

Dry, oil-free air is furnished throughout the plant by two 100-percent-capacity plant air compressors and one smaller, standby air compressor.

Personnel facilities consist of a personnel and maintenance building, two main control rooms (one in the RDF plant, one in the boiler-electric generating plants), a visitor's gallery, a chemistry laboratory, and restrooms in various areas. Virtually all process and control areas in the plant are housed in enclosed spaces that are either air conditioned or ventilated by fans and louvers.

CONCLUSION

The Palm Beach County resource recovery facility, and others like it, are part of the long range solution to solid waste management. The total solution involves the integration of recycling, resource and energy recovery, and landfilling.

NOTICE

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

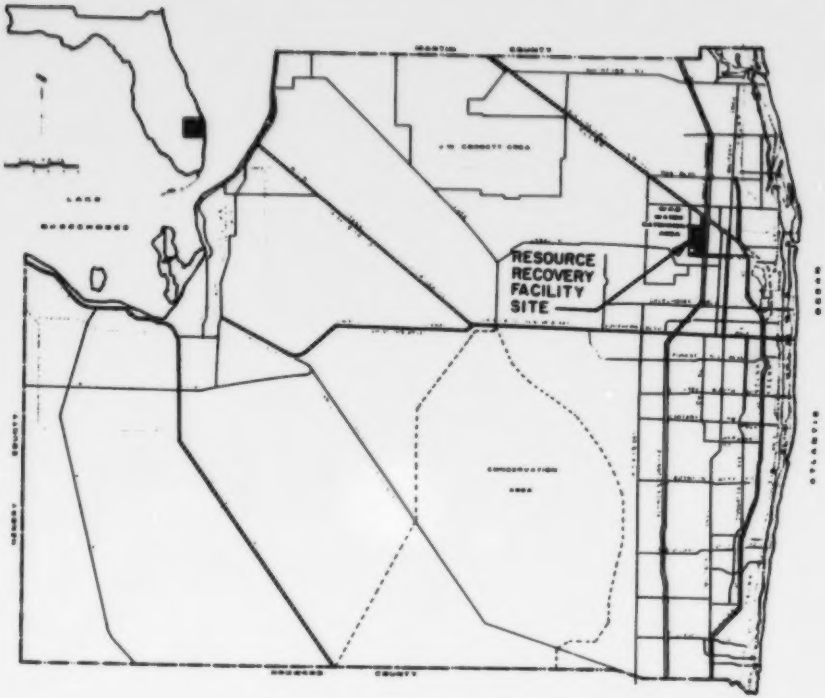

RESOURCE
RECOVERY
FACILITY
SITE

Figure 1: *Palm Beach County, Florida*

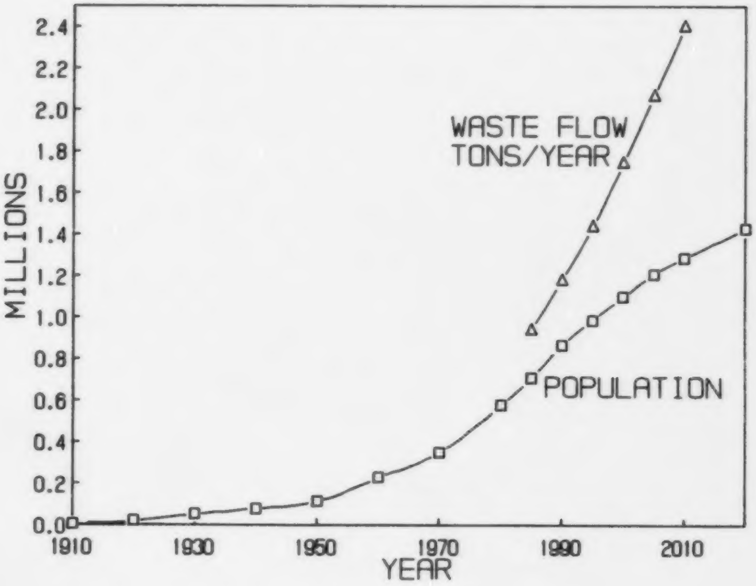


Figure 2: *Palm Beach County population and waste trends.*

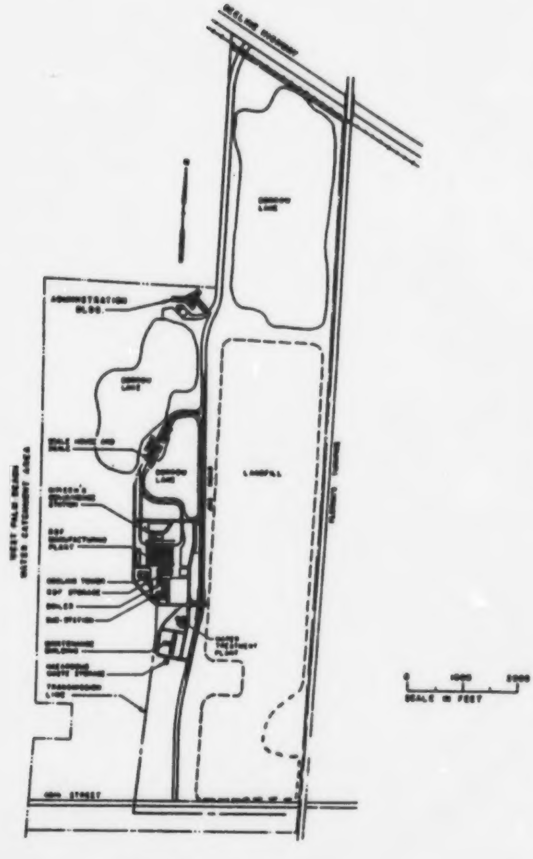

BEELINE HIGHWAY
ADMINISTRATION BLDG.
WEST PALM BEACH WATER CATCHMENT AREA
RDF MANUFACTURING PLANT
COOLING TOWER
RDF STORAGE
BOILER
SUB-STATION
TRANSMISSION LINE
LANDFILL
0 1000 2000
SCALE IN FEET

Figure 3: *Site and facilities.*

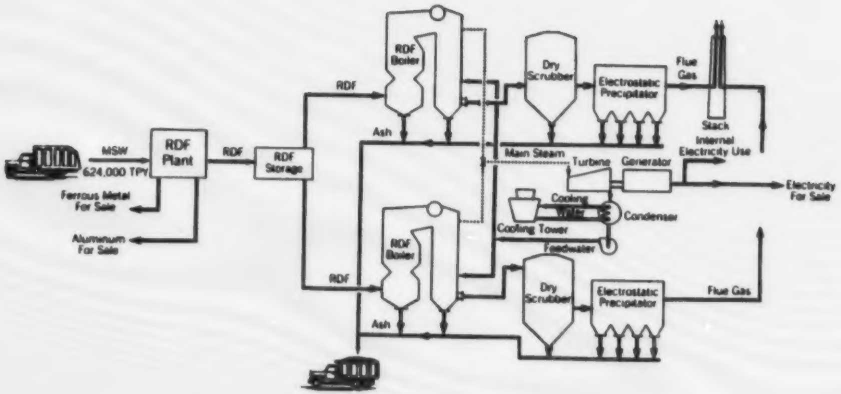

MSW
624,000 TPY
RDF Plant
Ferrous Metal For Sale
Aluminum For Sale
RDF
RDF Storage
RDF
RDF Boiler
Ash
RDF
RDF Boiler
Ash
Dry Scrubber
Electrostatic Precipitator
Flue Gas
Stack
Internal Electricity Use
Main Steam
Turbine
Generator
Electricity For Sale
Cooling Water
Condenser
Cooling Tower
Feedwater
Dry Scrubber
Electrostatic Precipitator
Flue Gas

Figure 4: *Facility flow diagram.*

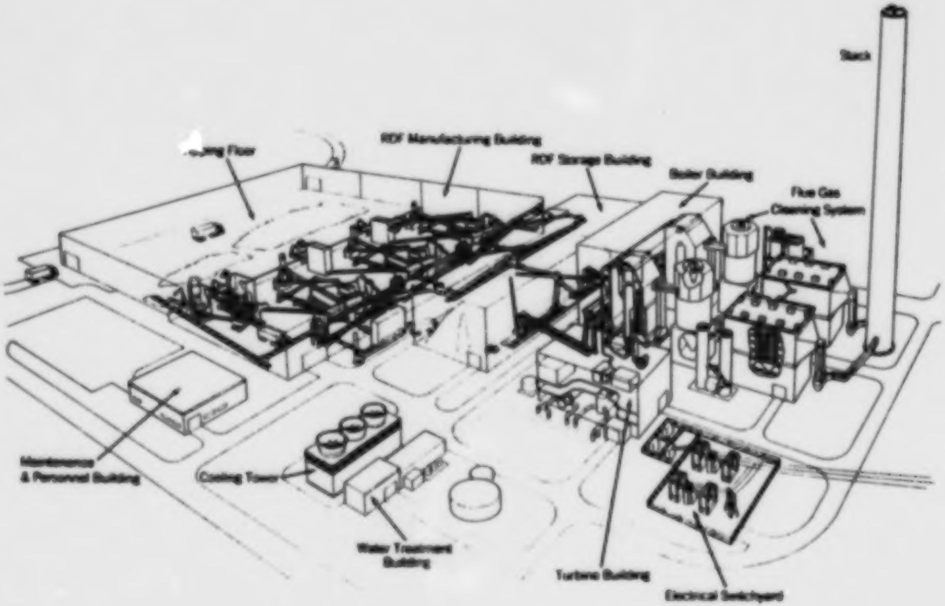
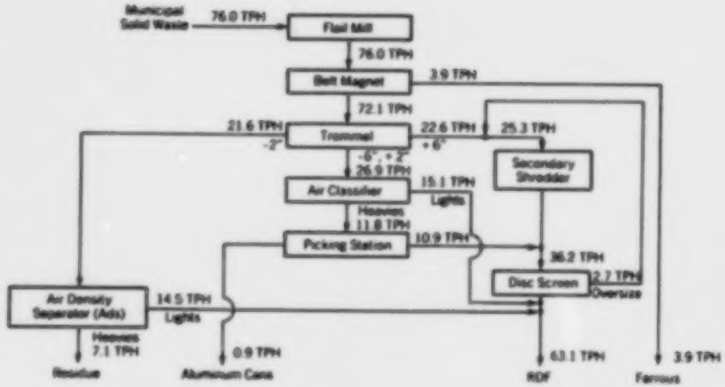

ping Floor
RDF Manufacturing Building
RDF Storage Building
Boiler Building
Flue Gas
Cleaning System
Stack
Maintenance
& Personnel Building
Cooling Tower
Water Treatment
Building
Turbine Building
Electrical Switchyard

Figure 5: *Facility layout.*


Municipal Solid Waste
76.0 TPH
Flail Mill
76.0 TPH
Belt Magnet
3.9 TPH
72.1 TPH
21.6 TPH
-2"
Trommel
22.6 TPH
+6"
25.3 TPH
-6", +2"
26.9 TPH
Secondary Shredder
Air Classifier
15.1 TPH
Lights
Heavies
11.8 TPH
Picking Station
10.9 TPH
36.2 TPH
Disc Screen
2.7 TPH
Oversize
Air Density Separator (Ads)
14.5 TPH
Lights
Heavies
7.1 TPH
Residue
0.9 TPH
Aluminum Cans
63.1 TPH
RDF
3.9 TPH
Ferrous

Note: Evaporative Water Loss Not Shown

Figure 6: *RDF plant mass flow diagram.*

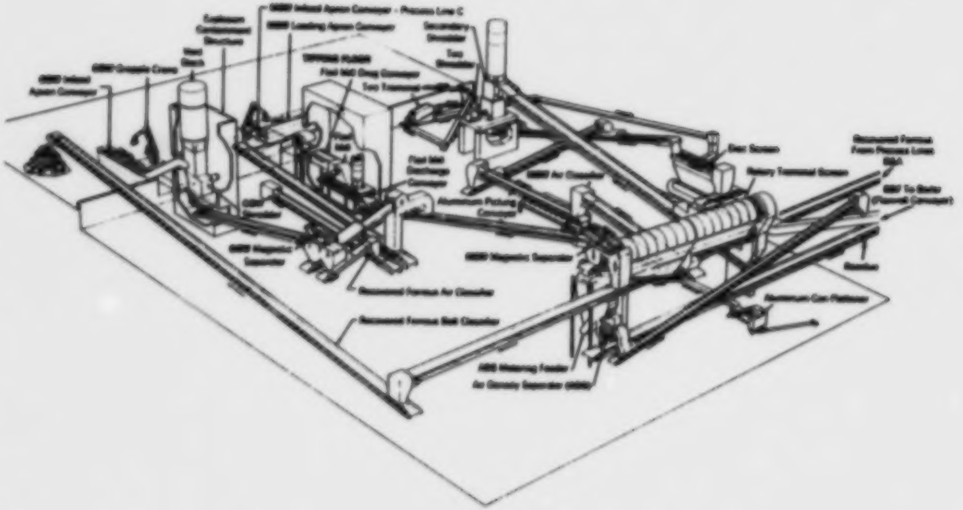

MSW Infeed Apron Conveyor
MSW Grapple Crane
MSW Infeed Apron Conveyor - Process Line C
MSW Leveling Apron Conveyor
Secondary Shredder
TIPPING FLOOR
Fluff Mill Drag Conveyor
Fluff Mill Discharge Conveyor
Aluminum Picking Conveyor
MSW Magnetic Separator
Recovered Ferrous Air Classifier
Recovered Ferrous Belt Conveyor
MSW Air Classifier
Disc Screen
Rotary Trommel Screen
Recovered Ferrous From Process Lines (B&C)
RDF To Baler (Transfer Conveyor)
MSW Magnetic Separator
Baler
Aluminum Can Flattener
ADS Metering Feeder
Air Density Separator (ADS)

Figure 7: *RDF plant.*

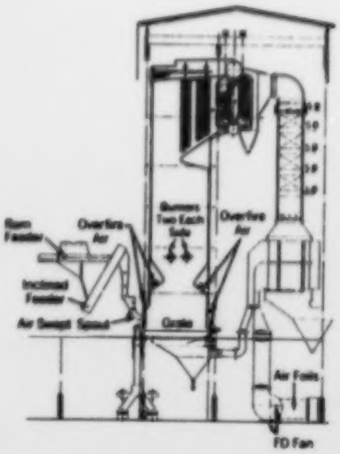

Ram Feeder
Overfire Air
Burners Two Each Side
Overfire Air
Inclined Feeder
Air Swept Spout
Grate
Air Foils
FD Fan

Figure 8: *State of the art CCZ RDF boiler design.*

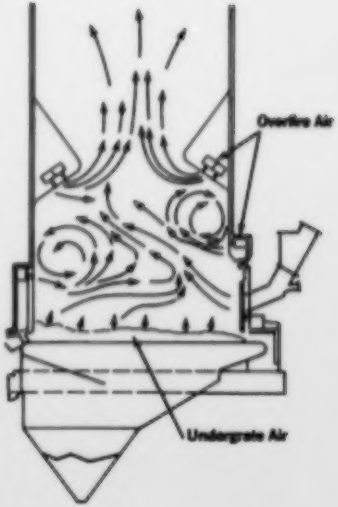

Overfire Air
Undergrate Air

Figure 9: *Controlled combustion zone (CCZ) lower furnace.*

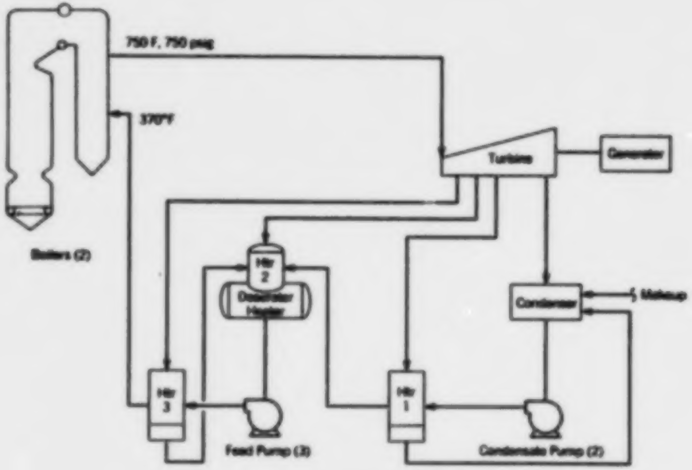

750 F, 750 psig
370°F
Boilers (2)
Turbine
Generator
Htr 2
Deaerator Heater
Condenser
Makeup
Htr 3
Feed Pump (3)
Htr 1
Condensate Pump (2)

Figure 10: *Steam turbine cycle.*

Table 1. Reference Waste

Component Analysis	Reference MSW (% by Weight)		Refuse-Derived-Fuel (% by Weight)	
Corrugated Board	5.46		-	
Newspapers	17.16		-	
Magazines	3.44		-	
Other Paper	19.46		-	
Plastics	7.24		-	
Rubber, Leather	1.94		-	
Wood	0.83		-	
Textiles	3.07		-	
Yard Waste	1.11		-	
Food Waste	3.71		-	
Mixed Combustibles	17.52		-	
Ferrous	5.43		-	
Aluminum	1.80		-	
Other Non-Ferrous	0.32		-	
Glass	11.51		-	
Total	100.00			
Ultimate Analysis				
Carbon	26.65		31.00	
Hydrogen	3.61		4.17	
Sulfur	0.17	(max. 0.3)	0.19	(max. 0.36)
Nitrogen	0.46		0.49	
Oxygen	19.61		22.72	
Chlorine	0.55	(max 1.0)	0.66	(max. 1.2)
Water	25.30		27.14	
Ash	23.65		13.63	
Total	100.00		100.00	
Heating Value	4728	Btu/lb	5500	Btu/lb
Fuel Value Recovery, percent of MSW			96	
Mass Yield, percent (lb RDF/lb MSW)			83	

LEACHING CHARACTERISTICS OF INCINERATOR RESIDUES AND POTENTIAL FOR MODIFICATION OF LEACHING

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ABSTRACT

Leaching behaviour of municipal incinerator residues is examined using a number of different extraction procedures. The influence of pH is examined and the results compared with the predictions from a geochemical speciation model. Despite the heterogeneous nature of incinerator residues, the leaching behaviour is found to be very systematic. Diffusion tube studies are applied to examine the behaviour of leaching metals in the waste matrices and in waste-soil combinations. The method proves to be a valuable tool for studying leaching behaviour at the waste-soil interface.

INTRODUCTION

For proper management of the ever increasing quantities of municipal incinerator residues, detailed information on the leaching behaviour of raw residues and of construction products containing these residues is needed. Several studies on the leaching properties of incinerator residues have been carried out (1-5). To date the results have been rather phenomenological, showing leach-rates, which very strongly with experimental conditions. These data are insufficient to predict the long term leaching behaviour of disposal sites and residue products. In this report, the leaching characteristics of incinerator residues are examined using a variety of experimental methods. Systematic behaviour and clear trends are observed in spite of the inherent variability of waste materials (1-3).

For a proper assessment of long term disposal and utilization options data obtained from leaching tests has to be complemented with information on the behaviour of leached in the soil. The change in soil porewater composition by percolation or diffusion of constituents from the waste affects the balance of major ions. Consequently, the retention capacity of the soil and the speciation of the waste constituents are altered. Diffusion tube methods provide a means to study these phenomena (6,7). This report discusses diffusion tube studies of incinerator residues, the consequences of these results for the interpretation of earlier data, and the practical implications for disposal and utilization of incinerator residues.

MATERIALS AND METHODS

MATERIALS

Sampling and sample preparation

Samples of incinerator slag and fly ash were obtained from AVR Waste Incineration Facility, a municipal solid waste facility with an annual capacity of 1 M tonnes. In the Rotterdam facility the slag is quenched and the iron fraction is removed magnetically. The slag is then sieved to a size less than 40 mm and stored in the open air. The water content in storage is approximately 16%(W/W). The sample for this study was acquired from the storage depot about two month after production. Prior to our experiments, the slag samples were quartered twice and dried at 60 °C. The slag was then crushed in a jaw mill (Retsch, GmbH) and subsequently in a rotary disk mill (Siebtechnik, GmbH) to obtain representative samples for analysis (95 % of mass below 125 μm). The fly ash was collected from electrostatic precipitator hoppers. The fly ash samples did not require pretreatment as the size distribution allowed representative sampling at the 0.5 - 1 g level.

Analytical methods

The chemical composition of the incinerator residues was analyzed using a number of methods. Na concentrations were measured by flame atomic absorption spectrometry (AAS) after fusion of the sample with $LiBO_2$ and subsequent dissolution in 1 N HCl. Ca, Zn, Cu, Pb, Cd were analyzed by flame AAS after the samples were dissolved in a mixture of concentrated acids (HNO_3 / $HClO_4$ / HF in the ratios 18:1:1). A solution of 1 N H_3BO_3 was used to mask the excess fluoride. Samples for As, Sb, Mo and Cr analyses were also treated with the strong acid and H_3BO_3 solutions. As and Sb were measured by hydride generation and AAS detection. Mo and Cr concentrations were measured by inductively coupled plasma emission spectrometry (ICP-AES). The extracts obtained from the leaching test were analyzed by the same methods. All extracts were stored after acidification to pH 2 with 1N HNO_3.

Leaching tests

To characterize the leaching properties of incinerator residues a number of different leaching test were applied. The tests were chosen to cover a wide range of liquid to solid ratios and to allow the measurement of pH effects on leaching. -

The "leaching availability" test, according to the Dutch standard NVN 2508 (8), consists of an extraction at a liquid to solid ratio of 100 (l/kg) at $pH = 4$ for 4 hours. The material for this test is crushed until 95% of the mass passes 125 μm. The method completely dissolves moderately soluble mineral phases, such as calcium and magnesium carbonates and sulfates.

The multiple batch extraction procedure, Dutch standard NVN2508 (8), consists of five subsequent extractions using the same starting material. The test is carried out with demineralized water at a liquid to solid ratio of 20 (LS = 20) in each cycle. The effect of untreated versus ground material on leaching was studied in a multiple batch extraction in 1 liter polyethylene flasks and in a 25 liter vessel. In the small container, both crushed incinerator slag and untreated material were tested (tests in duplicate). In the large container, only untreated material was tested.

Porewater was simulated by single extractions at liquid to solid ratios of 0.5 and by multiple batch extractions at LS values ranging from 4 to 0.5. For the multiple extractions, fresh solid was added to the same amount of leachant.

The leachability of incinerator residues as a function of pH was tested using similar methods as were applied in an earlier study of coal fly ash (9). In this case, the pH was adjusted with a 1:1 mixture of 1 N HCl and 1 N H_2SO_4 to obtain lower pH values than the equilibrium pH of the residues (pH 9-11). The pH was adjusted with the addition of CaO to reach higher values. To reach a pH of 12 a small quantity of 2.5 N NaOH was needed. These acids and bases were chosen because they are thought to be the pH controlling species in the unaltered wastes. The experiments with slag and fly ash were carried out at LS = 100 and at LS = 5, respectively. The pH was fixed for 4 hours and the solutions were stirred continuously. All extracts were filtered through 0.45 μm membranefilters (Schleicher&Schull) and acidified to pH = 2 for later analysis.

Diffusion measurements

The diffusion tube experiments consisted of mixing radiotracers with the incinerator wastes, placing the mixture into a tube, and filling the tube with unlabelled material. After the radiotracer diffused into the unlabelled compartment, the tubes were sliced into sections and analyzed (7). Two types of experiments were conducted. In the first, the non-labelled media was the same as the labelled, i.e., fly ash or slag. In the second, waste-soil interface effects were studied by placing soil materials in the unlabelled half of the tubes. All experiments were carried out under no-flow conditions at approximately 80 % water-saturation.

The physical retardation (tortuosity) in the diffusion tube was estimated from experiments with a non-interacting component. In most cases, Na-22 ($T_{1/2}$ = 2.6 y, E_γ = 1274 keV) was suitable for this purpose (10). The tortuosity factors were calculated as the quotient of the free diffusion coefficient in water and the effective diffusion coefficient measured in the tube. Diffusion profiles for interacting elements yielded information on chemical interaction effects. After correction for tortuosity, chemical retardation factors for each element were obtained from the ratio of the element's free diffusion coefficient to the measured effective diffusion coefficient. The isotopes studied were Cd-115m ($T_{1/2}$ = 44.8 d, E_γ = 336 keV), Mo-99 ($T_{1/2}$ = 66.6 hr, E_γ = 140 keV), Sb-125 ($T_{1/2}$ = 2.77 y, E_γ = 428 keV), As-74 ($T_{1/2}$ = 17.7 d, E_γ = 596 keV), Pb-210 ($T_{1/2}$ = 22.3 y, E_γ = 305 keV) and Zn-65 ($T_{1/2}$ = 244 d, E_γ = 1115 keV).

RESULTS

RESULTS OF ELEMENTAL ANALYSIS

Tables 1 and 2 show results of the elemental analysis on the incinerator slag and fly ash samples. Results are compared with literature data from Canadian (2), Danish (4) and other Dutch incinerators (1).

TABLE 1. COMPOSITION OF INCINERATOR SLAGS (mg/kg)

Element	This work		Ref.(1)	Ref.(4)	Ref.(2)
Na (%)	2.35	(0.09)	1.3 - 6.9	3.6	-
As	17.5	(0.4)	1 - 50	21	-
Cd	3.0	(0.05)	1 - 50	1 - 3	2 - 7
Cu	3780	(1450)	430 - 10000	2400 - 4300	850 - 26000
Cr	267	(48)	180 - 1500	230	900 - 1300
Mo	56	(8)	10 - 500	2 - 5	-
Ni	160	(96)	40 - 400	70 - 100	600 - 1300
Pb	2100	(1460)	1000 - 4650	2300 - 3100	1000 - 6000
Sb	56	(5)	1 - 600	-	-
Zn	2640	(110)	275 - 13400	2400	1400 - 3200

TABLE 2. COMPOSITION OF INCINERATOR FLY ASH (mg/kg).

Element	This work	Ref.(1)	Ref.(4)	Ref.(2)
Na (%)	2.53	0.34 - 1.2	2.3 - 2.9	-
As	1	5 - 80	50 - 70	-
Cd	237	10 - 300	250	400 - 1100
Cu	780	400 - 1400	900 - 1300	800 - 1400
Cr	715	200 - 1600	180 - 220	500 - 800
Mo	25	10 - 80	27	-
Ni	64	10 - 230	110	140 - 240
Pb	6090	800 - 13900	7600	11000 - 26000
Sb	297	20 - 600	-	-
Zn	14200	2000 - 21000	23000	25000 - 70000

Figures 1 and 2 and show the comparison as bar diagrams. Note the generally good agreement among the elemental composition of slags and fly ashes from different installations in different countries. The agreement is obtained in spite of the temporal variation in input to each incinerator.

The older data (1) show a much wider range than observed in recent studies. This may be due to improvement in sampling techniques, sample pretreatment, and analytical techniques.

In fly ashes, the elements showing significant differences between Canada and Europe are Cd, Pb and Zn. In slags, the elements Cu and Ni are significantly higher in Canada than in Europe.

Slags are inherently inhomogeneous with respect to Cu and Pb (11) because of the occurrence of solid metal particles. It appears to be impossible to control this source of variability. Its magnitude is evident from the analytical results on slags after extensive grinding and homogenization. The standard deviations for Cu and Pb are approximately 20 times higher than would be expected from analytical errors alone.

RESULTS OF THE LEACHING TESTS

Leaching availability

The leaching availability results indicate the fraction of each element tied up in poorly soluble mineral phases, i.e., the fraction that can not be leached under environmental conditions after complete degradation of the waste material. The availability for leaching is expressed as the leachable part of the total concentration (mg/kg). Leaching percentages from the availability test were found to be consistent between groups of comparable construction materials (12). Figure 3 shows the leaching availability "fingerprint" for incinerator slag.

TABLE 3. COMPARISON OF AVAILABILITY TEST DATA AND SEQUENTIAL CHEMICAL EXTRACTION DATA ON INCINERATOR SLAG (percent of total).

Element	Availability test(n=3)	SCE (2)
As	1.6 ± 0.4 %	-
Cd	69 ± 3.8 %	100 %
Cr	2 ± %	3 - 6 %
Cu	3.6 ± 0.3 %	3 - 6 %
Ni	9.1 ± 1.5 %	30 - 50 %
Pb	4.5 ± 3.2 %	10 - 40 %
Sb	3.9 ± 0.4 %	-
Zn	17.7 ± 0.2 %	30 - 40 %

In table 3, leaching percentages obtained with the availability test are compared with the results obtained by sequential chemical extraction (SCE)(2). Comparison with SCE data shows that the leaching percentages in the leaching availability test are lower than the combination of the first two steps of the SCE. The reproducibility of the availability test is good, with the exception of Pb; here again the inhomogeneous nature of incinerator slag is apparent. The sensitivity of the analytical method for Cd used in the SCE reference is too low for any conclusions to be drawn.

Simulated porewater composition

The composition of the simulated porewater is given in table 4. For comparison, data from field and column studies of incinerator slag disposal sites are included (4). The comparison indicates that the concentrations observed in the simulated porewater are reasonably consistent with field and column data.

TABLE 4. POREWATER COMPOSITION OF INCINERATOR SLAG (ppb).

	This work		Field	Column
Element	LS=0.5	LS=4 - 0.5	LS=0.01 - 0.4(4)	LS=0 - 0.5(4)
Na*	1950	2240	3000 - 7000	4800 - 5500
As	2.4	1.1	6 - 25	2
Cd	4.1	5.2	0.1 - 1	0.4 - 32
Cr	5	5	2 - 80	2 - 12
Cu	480	550	5 - 210	5 - 61
Mn	1530	580	-	-
Ni	100	70	5 - 2	0.7 - 1.6
Pb	5.5	6.5	5 - 19	1 - 6
Sb	31	39	-	-
Zn	220	420	10 - 320	20 - 320
pH	8.06	7.9 - 8.22	10.1 - 9.3	8.7 - 9.6

* Concentration in ppm.

Concentrations of some elements in the simulated porewater are higher than those observed in the field. This discrepancy can be partly attributed to the fact that the porewater simulation was carried out under oxidizing conditions, while the conditions in the incinerator disposal site are reducing (high sulfide level). The agreement between the column data (4) and the batch extractions is good with the exception of Ni. To predict leaching under reducing conditions, the laboratory tests carried out under oxidizing conditions are of limited use. Further methods to assess the significance of leaching under reducing conditions is needed.

Dissolved concentrations of Cu, Ni and Zn in the simulated porewater are also higher than predicted by the geochemical speciation model MINTEQA2 (13). An explanation might be the presence of organically complexed species. The speciation of elements leached from wastes is also fundamental to their behaviour in soil systems after elution from the waste.

Crushed versus untreated samples

Table 3 shows results from the test of sample preparation. Except for the occasional discrepancy, the reproducibility of the duplicates is better than 10-15 %. At low concentration levels, the analytical error becomes a limiting factor. For Na, As and Cu, the concentration levels at which the analytical uncertainty exceeds 10 % are 0.1 ppm, 0.5 ppb and 5 ppb respectively.

TABLE 3. REPRODUCIBILITY OF MULTIPLE BATCH EXTRACTION

Element LS ratio	1-liter container Crushed		1-liter container Untreated		25-liter vessel Untreated	
	Conc.	RSD* (%)	Conc.	RSD* (%)	Conc.	Overall RSD
Na (ppm)						
20	89.5	3	75.6	6	68.4	12
40	9.9	12	7.3	0.5	11.4	21
60	3.5	7	2.4	10	5.3	36
80	2.1	5	1.5	3	4.6	51
100	1.6	3	1.3	15	1.8	19
As (ppb)						
20	1.2	6	1.5	9	1.4	14
40	2.3	0.5	2.4	3	1.5	17
60	2.3	40	2.6	8	2.0	22
80	2.3	43	2.6	11	2.1	14
100	2.9	12	2.3	3	2.2	15
Cu (ppb)						
20	202	6	151	10	32	47
40	62	1	46	4	21	35
60	51	21	30	2	20	41
80	41	2	31	3	29	17
100	39	5	28	4	17	32
pH						
20	9.0		8.7		8.4	
40	9.1		9.0		8.5	
60	9.1		9.1		9.1	
80	9.3		9.3		9.1	
100	9.4		9.5		9.7	

* Relative standard deviation

Table 3 shows results from the test of sample preparation. Except for the occasional discrepancy, the reproducibility of the duplicates is better than 10-15 %. At low concentration levels, the analytical error becomes a limiting factor. For Na, As and Cu, the concentration levels at which the analytical uncertainty exceeds 10 % are 0.1 ppm, 0.5 ppb and 5 ppb respectively.

The relative standard deviation due to different particle size in the small and the large container varies from 12 to 50 %. The largest deviation in the first extraction cycle was observed for Cu in the large vessel. Size reduction does not explain the difference, as untreated material has been tested in the small container as well. Furthermore, there is no significant difference in pH between small scale and large scale experiments. The inherent inhomogeneity of incinerator slag with respect to Cu may have been responsible for these results.

The deviations of less than 50 % are considerably lower than might be expected on the basis of the increase in surface area caused by sample grinding. Apparently, the chemical composition of the leachate is dictated by the incinerator ash matrix and solubility controls are not strongly dependent on the amount of exposed surface area. Similar observations have been made within the framework of test development in the Netherlands (14). If these observations can be confirmed for a number of different incinerator ashes, crushing of incinerator ash can be used for standardized test procedures.

Effect of pH

Figure 4 shows the leachability of Cd, Zn, Pb and Cu from incinerator slag and fly ash as a function of pH. Similar pH relations have been observed for pulverized coal ash (9). The data are also in good agreement with the pH effects reported in the NITEP study (2). The leachability of trace elements from incinerator fly ash and incinerator slag is apparently very similar.

The geochemical speciation model MINTEQA2 (13) was used to calculate the predicted distribution between dissolved and precipitated phases as a function of pH. Total elemental concentrations were obtained from the pH = 4 extract of the incinerator fly ash (LS = 5). The results are also shown in figure 4.

The agreement between the actual measurements and the model is good for Zn. For Pb and Cd, qualitatively similar patterns are observed, but actual solubilities appear to be sensitive to the Cl concentration in the leachate. The difference between fly ash and slag leachability is at least partly due to the higher Cl content in the fly ash. The agreement between the observed and the predicted leaching behaviour of Cu is poor at pH 9 - 10. Since the measured solubility is higher than that predicted with the geochemical model, an anionic Cu-complex is apparently formed in the leachate. Candidate species are organic acids, which have been identified in leachates of some incinerator residues (3) and have the potential to dissolve Cu in the pH range 6 - 9. The presence of reduced copper (Cu + 1) would also lead to a higher solubility due to the formation of $(\mathrm{CuCl}_n)_{1-n}$ complexes. Further speciation research is needed to elucidate the nature of the soluble Cu-species.

Figure 5 shows the leaching behaviours of As, Sb, Cr, Mo and Na from incinerator fly ash and slag. As expected, the Na solubility is independent of pH. The leaching of Mo and As is not strongly dependent on pH. In the case of As leaching from slag, the decrease noted at high pH is similar to observations in coal fly ash leaching (9). Cr appears to have a maximum leachability at pH 10 - 11. The behaviour of Sb is also very similar to that in pulverized coal ash (9).

The consistency in the data for fly ash and slag for two different LS ratios indicates that, although incinerator ashes may be inherently inhomogeneous, their leaching behaviour appears to be very systematic. As in the coal fly ash studies, we are planning to extend the current measurements to a wider range of LS ratios and pH values to characterize the leaching properties of incinerator residues in more detail. Verification of the consistency of data for different installations will then be required.

RESULTS OF DIFFUSION TUBE STUDIES

To illustrate the diffusion tube results, the behaviour of two elements, Cd and Mo, in slag and slag-soil experiments will be discussed. In these experiments, the pH in the porewater was found to be 8.2 ± 0.2. The physical retardation studies with Na showed a pD_e value of 9.27, where $pD_e = -\log D$ effective. This value results in a calculated tortuosity factor of 2.45.

Chemical retardation in the waste matrix

The diffusion measurements with Cd-115m in incinerator slag showed minimal mobility, which is in agreement with the porewater data and the pH relation. The effective diffusion coefficient for Cd, obtained after a contact time of 500 hr, ranges from 11.5 to 12. A more accurate measurement would require a contact time of at least 90 days. The free mobility of Cd^{2+} in water (15) is $pD = 9.14$. After correction for the tortuosity a retardation factor of 90 - 290 is obtained. A minor fraction (0.4%) with a pD_e of 9.5 was observed, indicating the presence of Cd in a chemical form that is not interacting with the matrix. The concentration of this fraction in the porewater at the interface is estimated from:

$$C_{pw} = \frac{SP \cdot f \cdot C_s}{LS}$$

with SP the mobile fraction (-), f the availability for leaching, Cs the solid phase concentration (mg/kg) and LS the water content in the tube (ml/g).
The calculation yields 0.034 mg/l at $LS = 0.25$, which is in reasonable agreement with the measured porewater concentration of 0.006 mg/l at $LS = 0.5$. A change in the pH of the system will cause a change in the effective diffusion coefficient as indicated by the leachability-pH relations. At a pH of 4 this would lead to a chemical retardation factor of approximately 1.4.

For Mo a relatively high mobility was observed. The effective diffusion coefficient of Mo, obtained after a contact time of 117 hr, was $pD_e = 9.80$. The free diffusion pD for Mo is 9.00 (15). After correction for the tortuosity a retardation factor of 2.6 is obtained. The porewater concentration at the interface is estimated at 6.5 mg/l at $LS = 0.25$, which also in reasonable agreement with the measured concentration of 1.5 mg/l at $LS = 0.5$.

Incinerator slag/soil interaction

The combinations of incinerator slag with clay, sand and oxidized bottom-sediment did not produce an appreciable change in the Cd mobility. In figure 7 the diffusion profile for Cd in the incinerator slag/sediment combination is shown. The pH is not changed substantially in these combinations. This indicates, that under no flow conditions in the pH range 7.5 - 9, a minor fraction of Cd (%) is released from the slag. A small increase in the fraction with the highest mobility is noted, when slag and sediment are fully mixed. This effect may be caused by the presence of soluble organic acids in oxidized sediment (16).

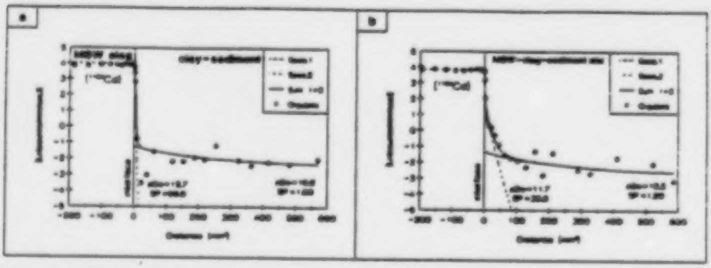

Figure 7. Diffusion of Cd incinerator slag into (a) clay-sediment and (b) an incinerator slag/clay sediment mixture (80/20 W/W). SP is fractionation.

Figure 8 shows the diffusion of Mo from slag into clay sediment. Note the complexity of the profile near the interface. Clearly, speciation and/or precipitation reactions have a strong influence on transport through this region. Similar results have been observed in other studies of waste-soil interfaces (7).

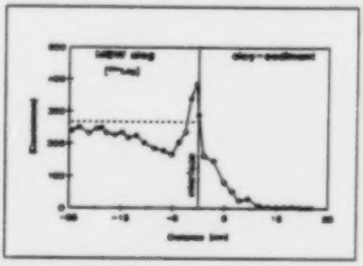

Figure 8. Diffusion of Mo from incinerator slag into clay-sediment.

DISCUSSION

The observed relationship between pH and leachability emphasizes the need for knowledge about long term pH changes in disposal or construction sites containing incinerator residues. Factors influencing the pH in the field are measures used during disposal, such as lime addition, and acidification due to biological activity (4,17,18). Self-neutralization effects, as observed in coal ashes due to pozzolanic reactions in the ash, are probably not of much significance in incinerator residues. More field measurements in existing, well-documented disposal sites are needed as input to further laboratory modeling studies.

The "safe" pH range for minimum leaching of metals is in the range from 8 - 10. Raising the pH above 10 by lime addition increases the potential for leaching of Pb, Zn, and Cu. At pH values below 7, a sharp increase in leachability of metals within one to two pH units is observed. These results suggest the need for guarantees that incinerator ash/slag disposals or utilizations do not fall below a pH of 7.

The above discussion applies only to oxidizing conditions. Under reducing conditions metals may be effectively retained as sulfides. Storage of incinerator residues in a permanently reducing environment might therefor lead to minimal metal release. For that purpose, mixing of incinerator residues with reducing sediments may prove beneficial. As shown by the porewater simulations, more work is needed to define leachability under reducing conditions.

Results from the studies of diffusion in waste matrices allow the flux of contaminants out of the matrix to be estimated. The retardation factor obtained with this procedure is based on the leachable fraction. Use of retardation factors based on the total elemental concentration would be misleading since the leachable fraction would migrate much faster and the peak concentration would be lower. The importance of aqueous speciation in determining chemical retardation is also apparent.

The diffusion tube method also shows the importance of interactions in waste/soil combinations. Differences in diffusivities and in porewater composition may lead to precipitation reactions at the interface. These effects are of particular interest on a macroscopic scale to assess compatibility of different wastes with proposed disposal scenarios.

CONCLUSIONS

The relationship between leachability and pH is indicative of the systematic leaching behaviour of incinerator residues . In some cases a good fit of laboratory experiments and geochemical modeling has been achieved. Further research into the chemical speciation of dissolved components may explain the discrepancies observed in other cases.

The composition of incinerator residues is encouragingly consistent for a wide range of samples. A few systematic deviations occur, such as the higher concentration level of Pb, Cu and Zn in fly ashes from Canada.

The composition of porewater extracts agrees reasonably well with field data. The effects of aqueous speciation of elements are most pronounced here, since dissolved concentrations exceed predicted dissolved levels.

The diffusion tube measurements allow quantification of chemical retardation at low LS ratios under conditions close to the field situation. In addition, interface effects can be studied. In case of incinerator slag, the mobility of Cd is low and that of Mo is relatively high. Retardation factors for the leachable fraction in incinerator slag for Cd are 90 - 300 and for Mo, 2.5.

For disposal and/or beneficial utilization of incinerator residues, knowledge of long term pH changes is important. The leaching of metals from incinerator residues becomes critical at pH values below 7. The pH-"window" for minimal metal impact ranges from 8 - 10.

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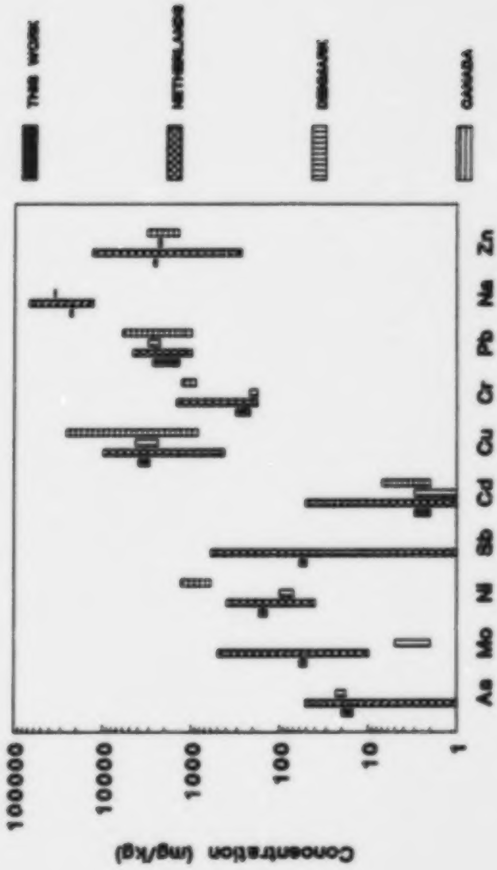

Figure 1. Fingerprint of incinerator slag composition.

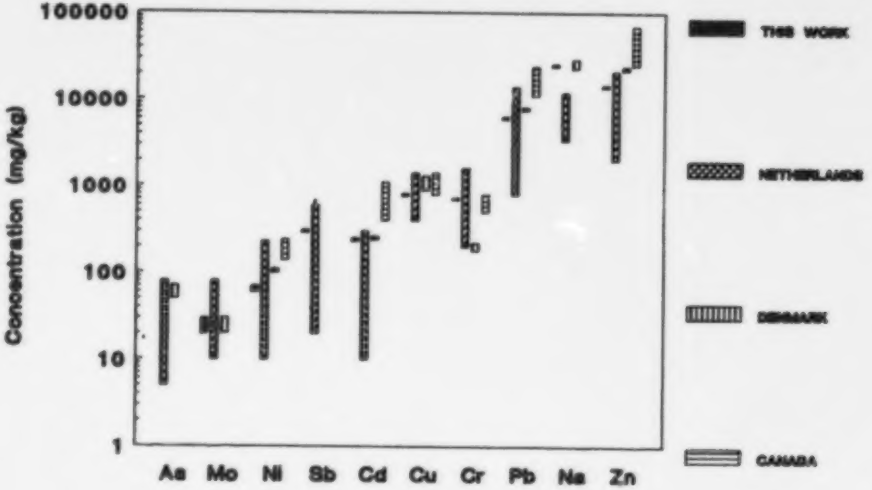

Figure 2. Fingerprint of incinerator fly ash composition.

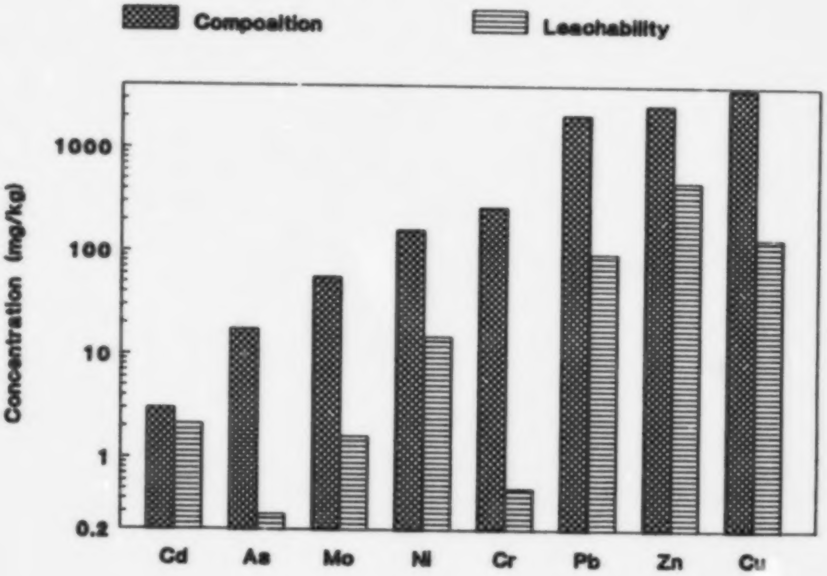

Figure 3. Composition and Leachability of MSW incinerator slag.

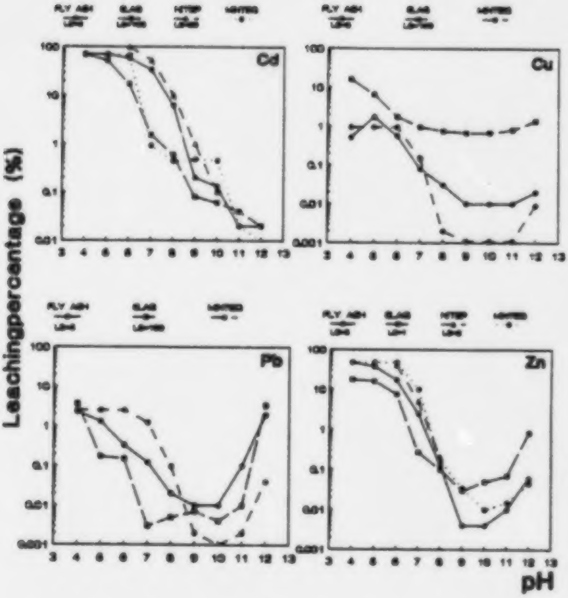

Figure 4. Leachability of Cd, Cu, Pb and Zn from MSW incinerator residues as a function of pH and modeling results using the geochemical speciation model MINTEQA2.

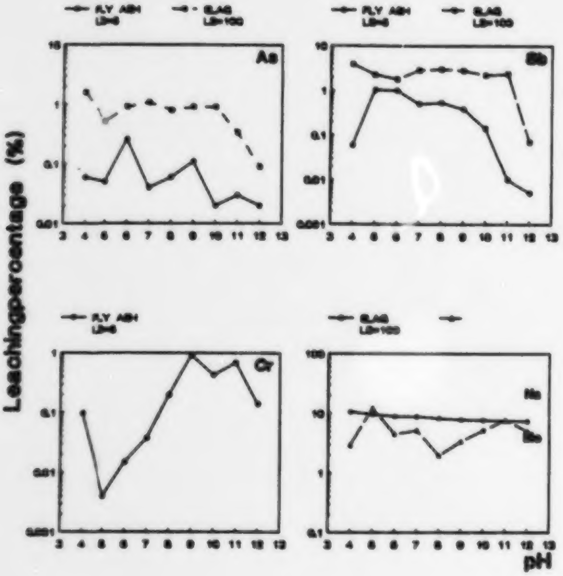

Figure 5. Leachability of As, Sb, Cr, Mo and Na from MSW incinerator residues as a function of pH.

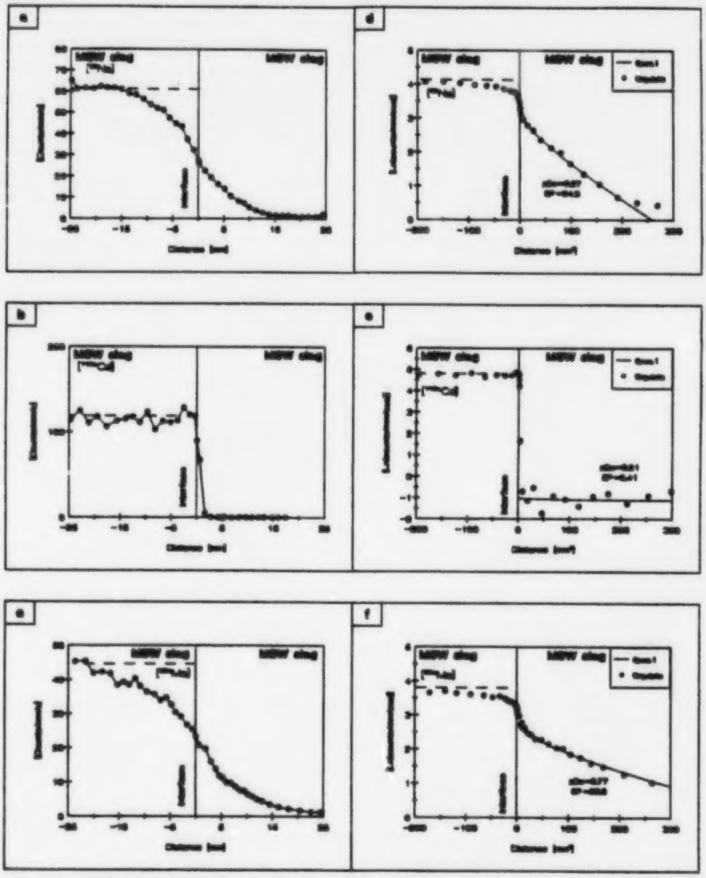

Figure 6. Diffusion tube profiles for Na, Cd and Mo in MSW incinerator slag and calculated effective diffusion coefficients.

EVALUATION OF pH AND OXIDATION-REDUCTION POTENTIAL ON LEACHABILITY OF MUNICIPAL SOLID WASTE INCINERATOR RESIDUES

BY

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ABSTRACT

Controlled batch experiments and a geochemical thermodynamic computer model (MINTEQ) were used to assess the influence of pH and oxidation-reduction potential (ORP) on metal solubilities of MSW incinerator residues. Batch experiments indicated that the pH has a significant influence on the aqueous phase concentration of Ca, Zn, Cd, Cu, Ni, Fe, Mn, Pb, and Al. The influence of ORP was significant for mobilization of Zn, Cu, Ni, Fe, Pb, and Al. Aqueous metal concentrations were higher for ash exposed to acidic reducing conditions compared to alkaline reducing conditions. A geochemical equilibrium model, MINTEQ, was able to mimic trends and changes in experimentally derived metal solubilities in batch extractions for a wide range of pH and ORP. Solid phases predicted to precipitate and thus control aqueous phase concentrations of metals were similar to ones observed with x-ray diffraction (XRD) for oxidized conditions. Under reducing conditions, many of the predicted solid phases (sulfide-based) differed from those identified by XRD; suggesting diagenesis or formation of new solid phases had occurred. There is potential to improve model predictions of leachate concentration at different L/S ratios. These results have implications for the management of incinerator ash. Environmental conditions, such as pH and ORP, affecting metal mobilization or immobilization may be evaluated without extensive laboratory testing programs.

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INTRODUCTION

Accurate characterization of leachable contaminants in municipal solid waste (MSW) incinerator residues prior to disposal is an integral component of overall waste management strategy in the United States (Francis et. al., 1988). Various laboratory techniques have been used to evaluate potential leachability of hazardous residual components, e.g. toxic metals, in a landfill environment. These leaching techniques can generally be classified as column or batch extractions (Jackson et. al., 1984).

There are inherent advantages and disadvantages to both column tests and batch tests. Column tests are often considered to be more representative of field conditions than batch tests (Cote, 1981; Constable and Cote, 1986; Hjelmar, 1987). However, column tests have been criticized because their duration can range from weeks to months and the results are difficult to reproduce (Jackson et.al., 1984; Constable and Cote, 1986). The lack of column reproducibility may be due to wall effects, channeling, non-uniform packing of waste, clogging and biological growth (Constable and Cote, 1986). Batch tests are comparatively rapid and noted for ease of operation. Batch test results are more reproducible than columns since conditions are more easily controlled by laboratory personnel. (Jackson et. al., 1984; Constable and Cote, 1986). This lower experimental variation should facilitate interlaboratory comparisons which are a necessary part of standardizing an extraction procedure for regulatory purposes (Jackson et. al., 1984). Consequently, most of the recent development work on leaching tests have been done by batch testing (Constable and Cote, 1986).

In general, speciation of metals in aqueous solutions containing MSW incinerator residues can be classified as:

- free ions,
- soluble complexes,
- adsorbed species, or
- solid phases/precipitates.

Chemical processes that influence metal speciation, and hence mobility, can include dilution, complexation with inorganic and organic ligands, precipitation-dissolution, adsorption-desorption and ion-exchange with inorganic and organic solids, and oxidation-reduction (Stumm and Morgan, 1981). The principal "master variables" in the aqueous environment that influence chemical speciation processes and the ultimate fate of metals include pH, oxidation-reduction potential (ORP), ionic strength (I), and temperature.

There are many environmental conditions present in a field disposal situation which are not easily measured and can not be routinely reproduced in a laboratory test (Constable and Cote, 1986). Some of the more notable problems have arisen in assessing microbial activity, metal speciation, oxidation-reduction (redox) equilibria, temperature, interactions among the waste material, and mass transfer and kinetic constraints.

The selection of a leaching media for a laboratory test is dictated by the intent of the test or the anticipated disposal scenario. Three landfill situations have been identified that could represent extremes in the pH of the leaching medium composition (Ham et. al., 1979). These include (i) a monofill situation where the leaching media would consist of an acid rain in the top layers of waste and of leachate produced from the waste in the bottom layers, (ii) a co-disposal situation with municipal refuse where the leaching media reflects the characteristics of MSW landfill leachate, and (iii) a co-disposal situation with other industrial waste where the pH of the leaching media would be waste specific. Further, ultimate disposal of ashes with organic wastes (or ashes by themselves) can produce environments that are also reducing because of microbial activity thus necessitating a need to understand the combined effects of pH and ORP on all types of residues.

The aggressiveness of the leaching medium results from its pH, redox potential, complexation capacity, and ionic strength (Constable and Cote, 1986). Numerous batch leaching tests have been evaluated to determine both their relevance to those factors that control leaching in actual landfills and their potential with respect to regulatory interpretation (Lowenbach, 1978). The parameters found to be most significant in the leaching of constituents from wastes were (i) pH of the extraction medium, (ii) liquid/solid ratio (L/S), and (iii) redox environment of the leach test.

The objective of the research presented herein was to evaluate the relative influence of pH and redox potential (ORP) on metal leachability of MSW incinerator residuals. The leaching of Fe, Mn, Pb, Cd, Al, Zn, Cu, Na, Ni, and Ca from a combined ash (8% fly ash and 92% bottom ash by volume) was emphasized. A two-level factorial design was used in controlled batch experiments (L/S 5) to determine relative importance of pH and ORP on selected metal solubilities. Qualitative pH and ORP influences on metal mobility were also evaluated by a computer model (MINTEQ) based on geochemical thermodynamic equilibria. Shifts in total soluble metal concentration within a pH range of 4 to 10 and ORP of -250mV to +600mV were mapped by computer model calculations. Minerals precipitated by model productions were compared with minerals observed with x-ray diffraction.

MATERIALS AND METHODS

Incinerator Ash Description

The combined ash used in the study originated from a community incinerator located in Durham, NH. The mass burn facility is comprised of three modular starved air incineration units (36 tons/day) and two low pressure steam generating boilers. The incinerators were manufactured by Consumat Inc. and the boilers manufactured by Vierson Boiler Works, Inc. The incineration takes place at 1400 to 1600°F under starved air, low turbulence conditions in the lower chamber. Hot gases are then reignited in the upper secondary chamber where excess air is added. Complete combustion occurs in the upper chamber at 1800°F.

The combined ash was mixed in the field at the time of sampling. The bottom ash was collected from a storage pile after exposure to leaching environments in the incinerator quench tank and in the storage pile. The fly ash was collected from the incinerator ductwork. The bottom ash was sieved to remove material greater than one-half inch, and mixed into a homogenous blend of 92% botton and 8% fly ash by volume. The collected ash typified normal operation of the incinerator facility.

Controlled Batch Experiments

The batch reactors were constructed from modified HDPE one-gallon containers as show in Figure 1. The reactors were filled with Type II water and ash at a 5:1 L/S ratio on a weight basis. Access ports were installed through PVC reinforced caps. A diffuser stone was installed near the reactor bottom to ensure even dispersion of nitrogen or oxygen gas bubbles and to provide periodic agitation of the reactor contents.

The batch reactor experimental design was configured in a two-level factorial matrix ($2x2^2$). The centerpoint was replicated four times to assist in the provision of an experimental lack-of-fit term to evaluate nonlinearity in the analysis of variance (ANOVA). Individual levels of pH and ORP that were tested are outlined in Figure 2. Seven replicated samples were conducted in an initial study at a given experimental condition (pH 10, high ORP, L/S 5) to determine sample and procedural variability. Aqueous phase concentrations of all the metal analytes had coefficients of variation of 5% suggesting that only duplicate batch runs need to be conducted at each experimental condition to provide a reasonable measure of detectable difference.

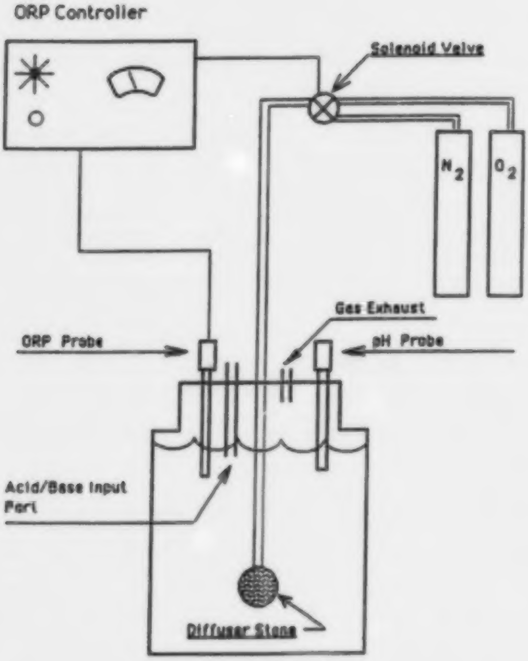
ORP Controller
Solenoid Valve
N2
O2
Gas Exhaust
ORP Probe
pH Probe
Acid/Base Input Port
Diffuser Stone

Figure 1. Schematic of experimental batch reactor

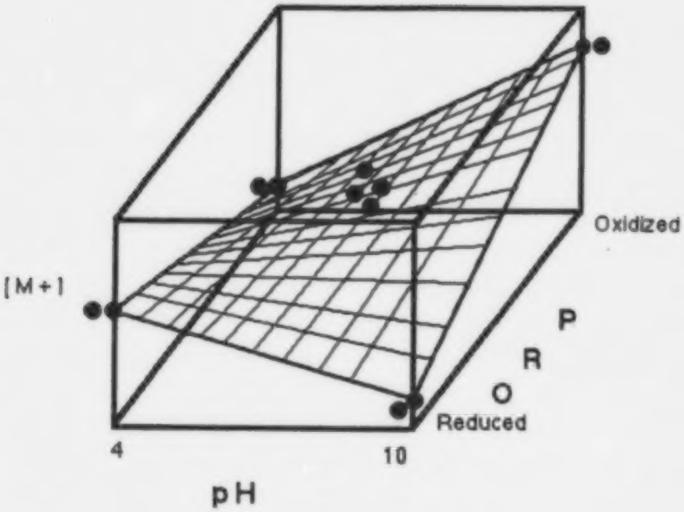
Oxidized
[M+]
P
R
O
Reduced
4
10
pH

Figure 2. Factorial design used to measure aqueous metal concentration as a function of pH and ORP

Batch pH and ORP were controlled by separate procedures. Small aliquots of HNO_3 or HCl and NaOH were added manually to the desired experimental level. Nitric acid was not used in those experiments involving reducing conditions since the nitrate could participate in redox reactions.

ORP was controlled by the method described by Patrick et al. (1973) with minor modifications. Oxidized conditions (high ORP) were obtained by diffusing oxygen into the system. Reducing conditions (low ORP) were achieved by seeding the reactor with 4% by weight domestic sewage sludge that has been characterized elsewhere (Eighmy et. al., 1988). Nitrogen was then bubbled into the reactors to purge the oxygen out of the system. Centerpoint ORP levels were achieved by first reducing the system as described previously and then slowly raising the ORP to the desired level by use of a ORP controller (Cole-Palmer; Chicago, IL) as shown in Figure 1. The controller could activate a solenoid value that would allow nitrogen or oxygen gas into the reactor as required to achieve the desired setting. The apparatus was able to consistently maintain the ORP to the target setting ± 10 mV. Stable oxidized conditions were typically achieved within 3-5 days while stable reducing conditions usually required 2-3 weeks, especially at the lower pH levels. It is thought that 2 to 3 weeks were required for the sludge-derived sulfate reducing bacterial populations to fully develop.

Once the appropriate pH and ORP were achieved, slurry samples were extracted from the reactors in a nitrogen purged glove box. The sample was withdrawn by a wide-bore plastic syringe and placed in a sealed Nalgene centrifuge tube. After centrifugation at 1000 rpm for 10 minutes, the centrate was filtered with a 0.45um filter (Nuclepore-Pleasanton, CA) and acidified. Samples were refrigerated until they were analyzed for elemental composition using inductively-coupled plasma (ICP) spectrometry. Analyses were performed by a commercial laboratory.

Modeling Strategy

The geochemical model MINTEQ is a third generation geochemical thermodynamic equilibrium model that uses the equilibrium constant approach to solve chemical composition problems (Brown et. al., 1986). MINTEQ is capable of calculating equilibrium aqueous speciation, adsorption, gas phase partitioning, solid phase saturation states, and precipitation-dissolution of metals. The foremost limitation of MINTEQ in simulations of actual landfill leaching phenomenon is due to the equilibria conditions implicit in the alogorithm. Equilibrium may be approached in batch experiments but dissolution, crystallization and adsorption reactions in lysimeters or in the field can be rate-limited by slow reaction kinetics or mass tranfer constraints.

The model strategy involved several distinct steps. The highest individual metal concentrations observed from the batch experiments were inputted to the model. The model was then executed using different pH/ORP combinations. Predicted leachate compositions in the output after solids precipitation could then be compared to the experimental results. An attempt was made to verify the model predicted solid phase compositon by x-ray diffraction of the original ash.

RESULTS AND DISCUSSION

Effects of pH and ORP

The pH and ORP levels used in the batch experiments were designed to achieve an orthogonal array necessary for statistical analyses. The upper and lower pH targets were selected to encompass pH levels reported in the literature for MSW incinerator bottom ash leachate (Hartlen and Elander, 1986;

Hjelmar, 1987; Eighmy et al., 1988; Thompson, 1989). The upper and lower target levels for ORP in this study were primarily determined by the oxidizing ability of oxygen and the reducing potential of 4 percent sludge seed under a nitrogen atmosphere. An achievable range around the ORP factorial categories of oxidizing, reducing and centerpoint conditions was ± 100mV.

As expected, pH levels were much easier to maintain within an acceptable tolerance (± 0.2) of the designated target than were ORP. Relative oxidizing and reducing conditions as measured by cell potential levels with respect to the standard hydrogen electrode, i.e. Eh, will vary roughly 59 mV for every unit change in pH. This equivalent change in relative ORP condition should parallel the boundary conditions of water. For example, an oxidizing and reducing condition of +400mV (Eh) and 0 mV (Eh), respectively, at pH 10 should equilibrate to ORP levels of +636mV (Eh) and +236mV (Eh), respectively, at pH 4. The center point for the orthogonal array should be at pH7 and +377mV (Eh). As shown in Table 1, all of the experimental conditions were close to acceptable tolerances of the targeted ORP level except for the low pH, low ORP condition.

Although the influence of low ORP at the low pH target of 4 was dampened, some interesting trends on the influence of pH and ORP on metal mobilities were apparent from the controlled batch experiments. Metal concentrations for each of the batch experiments at a specified pH and ORP condition are summarized in Table 1. Standard deviations are not shown because coefficients of variations for the replicate measures were less than 5 percent. The relative contribution of pH and ORP on metal leachability is described in Table 2.

The effect of pH produces the most noticeable changes in soluble metal concentrations. Except for Na and Al, metal concentrations decreased with

Table 1. Metal Concentrations as a Function of pH and ORP from Controlled Batch Experiments

Variables				Average Metal Concentration, ppm[b]									
pH	Targeted ORP	Actual ORP	Replicates	Ca	Na	Zn	Cd	Cu	Ni	Fe	Mn	Pb	Al
4	236	350	2	1740	850	114	1.22	0.32	0.53	39.5	35.4	1.43	51.5
4	636	600	2	1855	788	174	1.23	11.5	0.86	0.42	33.2	1.87	97.2
7	377	300	4	1323	822	8.5	0.24	0.11	0.22	BDL	20.0	BDL	0.05
10	0	0	2	277	1233	0.17	BDL	BDL	BDL	0.01	0.01	BDL	2.0
10	400	400	2	350	913	BDL	BDL	BDL	BDL	BDL	0.02	BDL	2.7

[a] ORP calculated as E_h in millivolts (mV) from platinum redox electrode.

[b] BDL - below detection limit for ICP, detection limits (in ppm) are Zn - 0.008, Cd - 0.009, Cu - 0.02, Ni - 0.008, Fe - 0.01, Pb - 0.066, Mn - 0.002, Ca - 0.1 and Al - 0.052

Table 2. Relative Influence of Variables on Metal Leachability in Controlled Batch Experiments.

	ANOVA Level of Significance[a]									
Variables	Ca	Na	Zn	Cd	Cu	Ni	Fe	Mn	Pb	Al
pH	**	-	**	**	**	**	*	**	**	**
ORP	-	-	*	-	**	*	*	-	*	*
pH * ORP	-	-	*	-	**	*	*	-	-	-
Lack-of-Fit	*	-	*	*	-	*	-	-	*	*

[a] (**)-Highly Significant, $p<0.01$; (*)-Significant, $p<0.05$; (-)-Not Significant, $p<0.10$.

increasing pH. In general, higher pH results in greater adsorption and precipitation while lower pH results in less adsorption and less precipitation (Theis and Richter, 1979). The Na concentration increased slightly at the higher pH possibly due to ion-exchange. The amphoteric nature of Al was reflected by the higher concentrations at pH 10 and 4 compared to the low value observed at pH 7. The pH of minimum solubility of Al hydroxides is typically shown to be near 6 (Amirtharajah and Mills, 1985).

The influence of ORP on metal solubilities was dramatic considering the constraints of the experimental conditions. The pH range used in the experiments varied over a much wider range of naturally available levels (and thus weighted toward extremes) compared to the experimental ORP range which occupied only 1/3 of the stability range of water, even less at the low pH level. Despite these limitations, ORP had a significant impact on soluble metal concentrations for Zn, Cu, Ni, Fe, Pb and Al. Except for Al, all of these inorganic species are known to precipitate as insoluble metal sulfides under reducing conditions (Lindsay, 1979). The ORP level necessary for sulfide induced precipitation could increase as the total aqueous sulfur concentration increases. Mn was not influence by ORP in this study possibly due to the limited ORP range at the upper and lower pH levels.

Variations in ORP may also affect metal mobilities by (i) directly changing redox sensitive metal oxidation states to more soluble/insoluble species, (ii) changing the extent of redox sensitive metal surfaces available for adsorption, and (iii) changing the degree of co-precipitation, precipitation, and complexation with other redox sensitive cations and anions.

A few metals in incinerator ash may be directly involved with redox coupled reactions. C,N,O,S,Fe, and Mn are predominant participants in aquatic redox processes (Stumm and Morgan, 1981). The Cu^{+1}/Cu^{+2} system may also be included as a direct participant in the redox reactions. Except for Mn as previously noted, Fe and Cu concentrations were directly affected by ORP by changes in oxidation states. Typically, the oxidized forms of Fe, Mn and Cu are more insoluble than the reduced forms.

The insoluble species of Fe and Mn formed from oxidized conditions may provide favorable adsorption sites for metals. Numerous researchers have observed the adsorption of Ni, Cu, Zn, and Cd by Fe and Mn oxides and oxyhydroxides (Jenne, 1968; Theis and Richter, 1979; Mott et al., 1987; Feijel et al., 1988). Iron oxides may also have some influence over those metals which form discrete precipitates such as Pb, Cr and Cu (Theis and Richter, 1979). The experimental results of this study suggest that of these metals only Cd was not measurably influenced by changes in ORP. The Fe oxide surfaces may have been primarily responsible for metals adsorption at the low pH low ORP conditions used in this study since most of Mn should have already been reduced (Gotoh and Patrick, 1972). Metal adsorption at the low pH, high ORP and higher pH conditions could be influenced by both Fe and Mn oxides.

Co-precipitation/precipitation with other cations and anions that change oxidation states may be significant in highly mineralized aqueous environments. Precipitation of metal sulfides or polysulfides in MSW incinerator ash under reducing aqueous conditions has been well documented (Eighmy et al., 1988). The sulfides of Pb, Fe, Ni, Cu and Zn may have been formed under the total sulfur concentration and low ORP conditions used in this study. Further chemical analyses are needed to verify the presence of sulfide solids. Aluminum concentrations may also be influenced by changes in oxidation states of sulfate as reflected by alunite ($KAl_3(SO_4)_2(OH)_6$) and $AlOHSO_4$. However, Al concentrations increased at the low pH, high ORP condition where no change or only a slight reduction was expected. This anomaly may be explained by an increase in aluminum-organic acid precipitates that can form after the addition of the organic sludge seed at the lower ORP conditions (Edzwald, 1986).

The influence of ORP is strongly affected by pH level as shown by the significant pH and ORP interactions listed in Table 2. Metal solubilization was more pronounced under conditions of low pH combined with low redox potential for Fe as reported elsewhere (Patrick et al., 1973) and low pH combined with high redox potential for Zn, Cu, Ni and Al. The lack of pronounced ORP influence at the high pH may have been due to low metal concentrations.

The lack-of-fit term listed in Table 2 is related to the nonlinearity of the response surface to the influence of pH and ORP on metal solubilities. Under the experimental conditions used in this study, Ca, Zn, Cd, Ni, Pb and Al developed nonlinear relationships between the variable extremes.

Model Predictions

A modified version of the metal speciation model MINTEQ was used to verify the metal solubility trends observed for the controlled batch experiments. The initial components to the model were the highest metal concentrations observed in the batch experiments as listed in Table 1. The same pH and ORP conditions controlled in the experiments were held constant during the computer runs. The resulting model predictions are shown in Table 3 and its corresponding statistical evaluations are summarized in Table 4. The modeling results indicate that metal solubilization was more statistically significant with respect to pH changes than to ORP changes, but both demonstrated relative statistical significance. Lower solubilities were observed at higher pH's and ORP extremes for most of the metals.

Table 3. Metal Concentrations as a Function of pH and ORP from Computer Equilibrium Model Predictions - L/S = 5.

Variables		Average Metal Concentration,[b] ppm									
pH	ORP[a]	Ca	Na	Zn	Cd	Cu	Ni	Fe	Mn	Pb	Al
4	330	305	1058	174	1.23	0	0.88	0.17	35.4	1.8	4.1
4	600	296	1058	174	1.23	11.5	0.88	0	35.4	1.8	3.9
7	300	300	1058	174	1.23	0	0.88	0	35.4	1.3	0.03
10	0	284	1058	0.12	0.03	0	0	0	0.16	0	0.02
10	400	288	1058	0.12	0.03	0.05	0	0	0	0	0.02

[a] ORP = E_h in millivolts (mV).

[b] Zeros indicate values below analytical detection limits

Table 4. Relative Influence of Variables on Metal Leachability in Computer Equilibrium Model Predictions.

	ANOVA Level of Significance[a]									
Variables	Ca	Na	Zn	Cd	Cu	Ni	Fe	Mn	Pb	Al
pH	-	-	**	**	**	*	*	**	**	**
ORP	-	-	-	-	**	-	*	-	-	-

[a] (**)=Highly Significant, p<0.01; (*)=Significant, p<0.05; (-)=Not Significant, p<0.10.

A comparison between the results from the batch experiments and from the model predictions reveal similar metal solubility trends especially at the upper and lower pH levels. The model predicted lower values than batch experiments for Al, Fe and Ca at pH 4 and a lower value for Al at pH 10. The greatest variability between experiments and computer model were observed at pH 7. The model predicted higher concentrations for Pb, Mn, Ni, Cd, Zn and Ca than what was observed in the laboratory at neutral pH. These differences may be explained by documented evidence that all of these metals can adsorb to iron and manganese oxide surfaces (Jenne, 1968; Williams and Patrick, 1973; Theis and Richter, 1979; Mott et al., 1987; Feijel et al., 1988). Incorporation of adsorption species into future MINTEQ runs may lower resultant metal concentrations closer to values analytically observed in the laboratory.

The mineral solids allowed to precipitate by MINTEQ were compared with minerals observed by x-ray diffraction. The comparison is summarized in Table 5. The minerals observed by x-ray diffraction were diverse (Eighmy et al., 1989). Most of the solids observed by x-ray diffraction include Ca and Fe aluminosilicates and other well defined crystalline structured minerals. Similar classes of minerals were observed between those found with x-ray diffraction and those predicted to form under oxidizing conditions. The limitations of x-ray diffraction for complete mineralogical analysis of incinerator ash has been attributed to the highly amorphous, non-crystalline nature of the ash (Henry et al., 1983; Ontiveros, 1988). The metals calculated by the model to precipitate were predominately sulfides under reducing conditions and sulfates minerals at the more oxidizing conditions.

The potential exists to improve model leachate predictions by varying the input recipe to produce more precipitated solids found in incinerator ash

Table 5. Comparison of Minerals Observed in Combined Ash Versus Precipitated Mineral Solids Predicted by Computer Equilibrium Model.

Minerals Observed By X-Ray Diffraction	Model Predicted Mineral Solids	
	Oxidizing Conditions[a]	Reducing Conditions[b]
$CaCO_3$	Fe_2O_3	Cu_2S
$PbSO_4$	$BaSO_4$	FeS
$CdSiO_3$	$KAl_3(SO_4)_2(OH)_6$	$AlO(OH)$
$Pb_3(PO_4)_3Cl$	$CaSO_4$	CUS
$FeCr_2O_4$	$Al(OH)SO_4$	Zn, $Fe(s)$
$NaAlBr_4$	$AlO(OH)$	PbS
$CdSO_4$		CdS
$CuFe_2S_3$		NiS
$CaSO_4$		$CuFeS_2$
$(Fe, Mg)SiO_3$		

[a] pH4 and + 600mV ORP; pH10 and + 400mV ORP.

[b] pH4 and + 350mV ORP; pH and 0mV ORP.

mineralogy studies. However, a discrepancy between analytically derived and computer generated minerals may exist where new solids may develop after exposure to a leaching environment which may not be found in the original ash.

The results of the MINTEQ computer simulations for Fe and Zn are presented graphically in Figures 3 and 4, respectively. The negative logarithm of the molarity is plotted as a function of pH and ORP, thus metal concentration in the leachate increases at the lower values on the vertical axis. Conversely, minimum solubility regions are denoted as the apparent "peaks". Deviations from classical theoretical solubility relationships could be due to the complexity and heterogeneity of the component input. A maximum leachate concentration "trough" is easily discerned from Figures 3 and 4, extending from pH 4 through 10 at approximately 0-50mV for both metals.

These results have implications for the management of ash piles and landfills. If in-situ leaching of metals for metal recoveries are required, then certain conditions promoting neutral to acidic pH (pH 4 to 7) and moderate ORP conditions (-100 to +100 mV) can be used to promote metal mobilization. If ash management practices are required to immobilize metals in piles or landfills, then conditions promoting neutral to basic conditions (pH 5 to 9) and very reducing or oxidizing ORP conditons (-300 to -100 mV, +100 to +400 mV) can be used to promote metal immobilization. These windows are metal-dependent.

CONCLUSIONS

Understanding the leachability of MSW incinerator ash exposed to varying environment conditions will provide valuable insight into sound management decisions. The influence of pH and ORP on metal mobilities in a combined ash is significant. The use of computer models to assess the leachability of ashes cannot yet be totally explained from thermodynamic

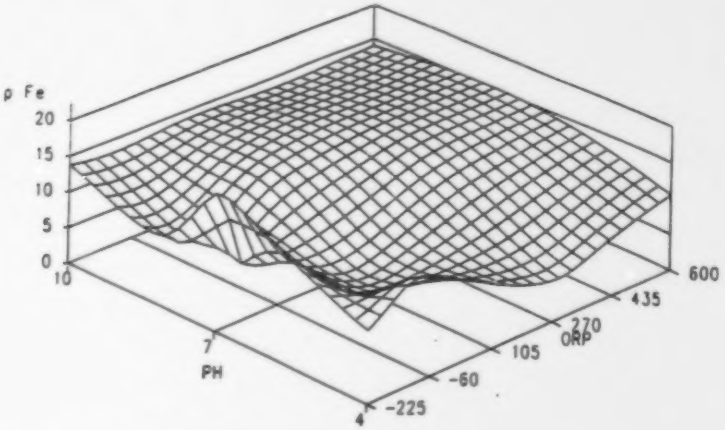

Figure 3. p [Iron] as a function of pH and ORP
from MINTEQ computer model simulations

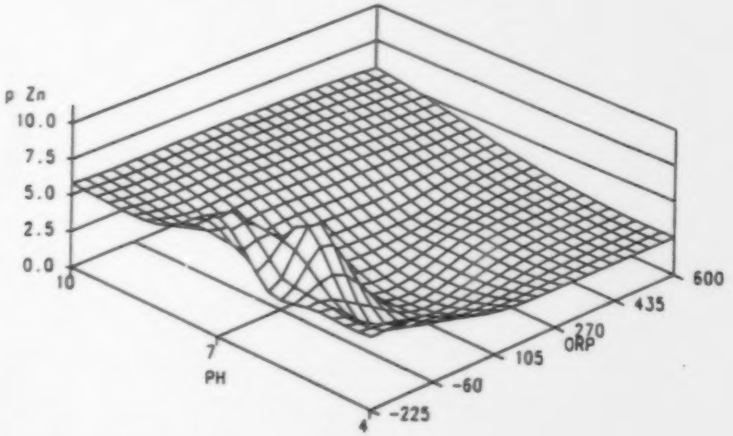

Figure 4. p [Zinc] as a function of pH and ORP
from MINTEQ computer model simulations

equilibrium considerations alone. However, the results obtained to date may be applied at a regulatory level as an initial indicator on the prognosis of an individual site or disposal scenario. The following conclusions can be drawn from the research presented herein:

- The effect of pH produces the most noticeable changes in metal solubility with most metal concentrations decreasing with increasing pH.
- The influence of ORP on metal solubilities can be selected but significant.
- Variations in ORP may affect metal mobilities by several mechanisms.
- The influence of ORP is strongly affected by pH and may be non-linear across extreme upper and lower pH levels.
- Chemical equilibrium model (MINTEQ) predictions were able to mimic experimentally derived metal solubilities. There is potential to improve model predictions.

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THE NATIONAL INCINERATOR TESTING AND EVALUATION PROGRAM: CHARACTERIZATION OF RESIDUES FROM A REFUSE DERIVED FUEL COMBUSTION SYSTEM AND A MODULAR MUNICIPAL WASTE INCINERATOR

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INTRODUCTION

In 1984, Environment Canada established the National Incinerator Testing and Evaluation Program (NITEP) to examine the impact of municipal solid waste (MSW) incineration on the environment. Under NITEP, the Wastewater Technology Centre initially evaluated the chemical properties and leachability of residues collected from two different types of MSW incinerator systems and an air pollution control pilot plant, using a battery of laboratory tests (Sawell et al., 1987 and 1988, Sawell and Constable, 1988). It became evident from these and other studies that the type of incinerator design, combustion conditions and air pollution control technologies, had a major impact on the chemical properties and leachability of the residues. As a result, the scope of the NITEP Ash Characterization Program was expanded to include the evaluation of ashes from several other MSW incinerators in Canada and the United States, to increase the understanding of contaminant behaviour in incineration and their fate under different residue disposal scenarios. This paper summarizes two studies in the expanded program, which evaluated various ash streams from a coarse refuse derived fuel combustion system with fabric filter air pollution control and a modular (two-stage) incinerator with lime-based air pollution control. Complete details of these studies are given in Sawell and Constable (in preparation).

Refuse Derived Fuel Combustion System

The Hamilton-Wentworth Solid Waste Reduction Unit (SWARU) is located in Hamilton, Ontario, Canada. It is classified as a refuse derived fuel (RDF) or semi-suspension incinerator facility, with coarse refuse pre-processing, including shredding and ferrous metal separation. In 1986, the facility underwent a major retrofit of the refuse processing, process control and air pollution control systems, to increase efficiency and reliability, and to reduce atmospheric emissions. The facility operates five days a week and is capable of handling 550 tonnes of refuse per day. Ferrous recovery accounts for approximately 5% (by weight) of the incoming waste stream.

A schematic of the facility is given in Figure 1. Pre-processed refuse is blown into one of two spreader stoker waterwall boilers. The refuse falls onto a moving grate which transports the ash across the bed of the incineration chamber before falling into a quench tank. Flue gases from the incineration chamber pass up through the

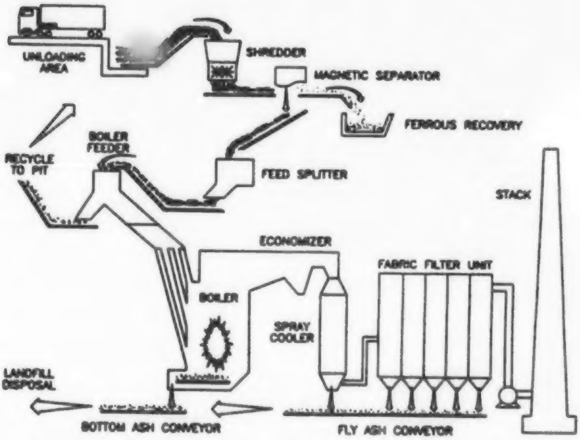
UNLOADING AREA
SHREDDER
MAGNETIC SEPARATOR
FERROUS RECOVERY
RECYCLE TO PIT
BOILER FEEDER
FEED SPLITTER
STACK
ECONOMIZER
FABRIC FILTER UNIT
BOILER
SPRAY COOLER
LANDFILL DISPOSAL
BOTTOM ASH CONVEYOR
FLY ASH CONVEYOR

FIGURE 1. CROSS SECTION SCHEMATIC OF THE SWARU FACILITY

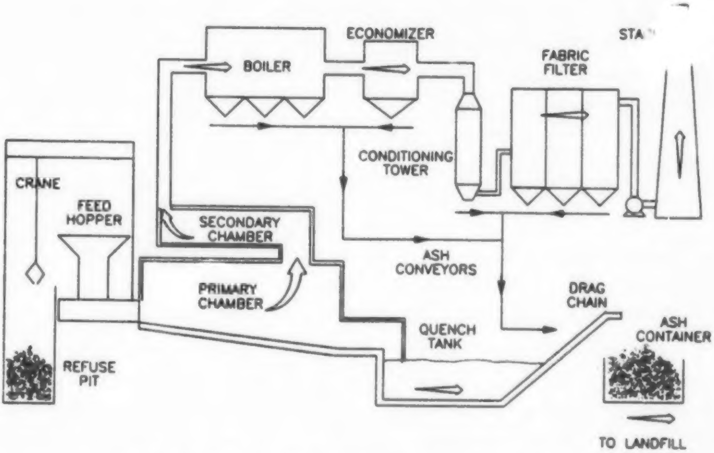
ECONOMIZER
STA
BOILER
FABRIC FILTER
CONDITIONING TOWER
CRANE
FEED HOPPER
SECONDARY CHAMBER
PRIMARY CHAMBER
ASH CONVEYORS
DRAG CHAIN
QUENCH TANK
ASH CONTAINER
REFUSE PIT
TO LANDFILL

FIGURE 2. CROSS SECTION SCHEMATIC OF THE LVH-EFW FACILITY

waterwall boiler and economizer heat recovery units into a spray cooling tower where the flue gases are humidified and cooled. The gases then pass through the fabric filter air pollution control unit prior to exiting the stack to the atmosphere.

Modular (Two-stage) Incinerator

The London Victoria Hospital Energy-From-Waste Facility (LVH-EFW) is located in London, Ontario, Canada. It consists of three parallel modular or two-stage incinerators with a lime-based air pollution control system. The facility has been in full operation since early 1988 and is designed to handle a total of 270 tonnes of refuse per day. No medical or pathological waste is processed through the facility.

A schematic of the system is given in Figure 2. Unprocessed refuse is fed into the oxygen starved primary combustion chamber via a loader ram. Transfer rams move the refuse down the bed of the chamber and the resulting ash is discharged into a quench tank, which also acts to seal the unit. Typically, the temperature in the primary chamber is just above 800°C. The fuel rich flue gases produced from this semi-pyrolytic process are then passed into a highly oxidative, high temperature (greater than 1000°C) secondary combustion chamber. The combusted flue gases then pass through the boiler and economizer heat recovery units before entering the spray conditioning tower, where they are humidified and cooled by a fine water spray. Dry lime is then injected into the gas stream prior to it passing through the fabric filter dust collectors.

SAMPLE COLLECTION

RDF System

Separate samples of bottom ash and fabric filter ash were collected over a 4 hour operating period under basic design operating conditions. Grab samples of bottom ash were collected every 20 minutes directly from the finishing grate, just prior to the quench tank. After coning and quartering, randomly selected quarters were composited. The composite sample was then cored using a 10 cm diameter soil sampler, and passed through a 9.5 mm sieve prior to testing and analyses.

Fly ash was collected on a periodic basis over the test period by diverting the fly ash from the conveyor leading to a pug mill, into a 200 litre drum. Each drum of fly ash collected was cored using a 2.5 cm diameter soil sampler and a composite sample of all the cores was mixed with a Brinkman PT Sample Mixer.

Modular Incinerator

Separate samples of bottom ash, boiler ash, economiser ash and conditioning tower/fabric filter (CT/FF) ash were collected over a five day period under design load operating conditions. Prior to sampling, the incinerator was shut down for maintenance and cleaning. In addition, the quench tank was drained, cleaned and filled with fresh water. The screw conveyor leading to the quench tank drag chain unit was blocked off to prevent fly ash contamination of the quench tank.

Grab samples of quenched bottom ash were collected from the ash storage bin once every two hours. The grab samples were composited daily, and cores (10 cm in diameter) from each daily sample were then dried at 40°C, mixed into one sample and sieved using the same procedure described above.

Samples from the boiler and economizer were collected by shutting down the screw conveyors leading from the respective hoppers during the test period, which allowed the ash to build up in the hoppers over the five days. At the end of the test period, the conveyors were turned on and each ash stream was diverted into separate 200 litre drums. Since small quantities of these ashes were collected, samples were obtained by taking cores using the sample jars. Based on the results from a previous NITEP study on a similar facility (Sawell et al., 1988), these ashes are considered to be fly ashes.

Combined ash from the CT/FF units was collected continuously over the five day test period by diverting the ash into 200 litre drums. One core (7.5 cm in diameter) from each full drum was placed in a separate container. In order to obtain a single sample, each of the containers were cored using a 1.5 cm diameter glass tube and the cores were composited and mixed using a Brinkman PT Sample Mixer. Since the lime injection ports malfunctioned during the last day and a half of testing, only the CT/FF samples from the first three and a half days were used for test purposes.

ANALYTICAL AND TEST METHODS

Ash samples from both facilities were analysed for their total metal concentrations using standard analytical techniques. Prior to digestion with aqua-regia and hydrofluoric acid, the ashes were dried and then ground to pass through a No. 100 mesh sieve. The metals concentrations in the ash samples and the leachates were measured using directly coupled argon plasma (DCAP), flame atomic absorption (AA) or AA with graphite furnace.

The fixed solids content of the ashes was determined based on the weight loss of a dry sample after 1 hour at 550°C (APHA Standard Method 209E).

The ashes from both facilities were subjected to the Sequential Batch Extraction Procedure (SBEP), ASTM Method No. D4793-88, which involved mixing the ash with distilled water at a 20:1 liquid-to- solid ratio for 18 hours. After mixing, the solid and liquid phases were separated via filtration through 0.45 micron pore size membrane filters. The solids were then re-exposed to fresh distilled water at the same liquid-to-solid ratio and the procedure was repeated for 5 cycles. The filtered leachates were preserved with nitric acid to a pH of 2 prior to analyses for metals. A 100 mL sub-sample of leachate was analysed for conventional parameters (e.g., pH, total dissolved solids, anions) without prior preservation.

The ashes from both facilities were also subjected to the Sequential Chemical Extraction Procedure (SCE) (Fraser and Lum, 1983). The SCE involved a step-wise separation of the total mass of a metal in each ash into five fractions. A summary of the SCE is given in Table 1.

Table 1. Summary of the Sequential Chemical Extraction Procedure

	Digestion Method
Fraction A	1 M LiCl/CsCl in 60% CH_3OH
Fraction B	1 M CH_3COONa with Acetic Acid
Fraction C	1 M NH_2OH + HCl in 25% Acetic Acid
Fraction D	Nitric Acid with Peroxide, Heat + CH_3OOH_4 in 20% Nitric Acid
Fraction E	Aqua regia/HF with peroxide + HCl

RESULTS AND DISCUSSION

Metal Concentrations

Since cadmium, lead and zinc are the major toxic metals of concern with regard to MSW incinerator ash, the discussion in this paper is limited to these metals. The average concentration of these metals in the ashes from both facilities are shown in Table 2.

Table 2. Metal Concentrations in the Ashes (ppm)

	SWARU Ashes		LVH-EFW Ashes			
Parameter	Bottom	Fly	Bottom	Boiler	Economizer	CT/FF
Cd	18	92	<5	788	763	821
Pb	1320	3200	1970	23800	24000	13400
Zn	4620	5530	6280	40300	35000	38500

Cadmium, lead and zinc were measured in higher concentrations in the fly ashes than in the bottom ashes. Based on the relative volatility of these metals under the temperatures inherent in a municipal waste incinerator, these results were not unexpected and are consistent with results from previous studies (e.g., Brunner and Monch, 1986; Vogg, 1987; Sawell and Constable, 1988).

The bottom ash from SWARU contained more cadmium (18 ppm) than the bottom ash from the LVH-EFW facility (less than 5 ppm). Conversely, the bottom ash from SWARU contained less lead and zinc than the LVH-EFW ash. All three types of fly ash from the LVH-EFW facility contained substantially higher concentrations (at least 4 times higher) of cadmium, lead and zinc than did the fly ash from SWARU.

The large difference between the concentrations of the three metals in the fly ashes from the two facilities is probably due to a combination of factors including: 1) the ferrous metal removal process at SWARU eliminated a portion of the cadmium, lead and zinc from the refuse feedstock, resulting in lower concentrations in the fly ash, and 2) the SWARU semi-suspension process causes much higher particulate loading to the air pollution control unit than that achieved in the modular incinerator, which results in a greater mass of non-metallic material and lower metal concentrations in the SWARU fly ash.

Fixed Solids Content

The fixed solids contents of the ashes from both facilities are given in Table 3. The bottom ash from SWARU contained about 95% fixed material, while the LVH-EFW bottom ash contained only about 71% fixed solids. This large difference can be explained by the fact that semi-pyrolytic conditions in the primary chamber of a modular incinerator (LVH) are much less efficient in combusting material than a speader stoker unit, especially when the refuse is shredded and blown into the furnace, as done at SWARU.

Table 3. Fixed Solids Content of the Ashes (% dry weight)

	SWARU Ashes		LVH-EFW Ashes			
	Bottom	Fly	Bottom	Boiler	Economizer	CT/FF
Fixed Solids	95.1	89.4	70.8	99.9	97.8	94.1

The fixed solids content of bottom ash from a modular incinerator examined during a previous NITEP study (Environment Canada, 1985) generally ranged between 80 and 85%. The much lower fixed solids content in the bottom ash from the LVH-EFW facility suggests that the incinerator was not operating under optimum conditions. The high combustibles content of ashes from modular incinerators may preclude utilization of this material without prior processing.

The fixed solids content of the fly ash from SWARU was about 90% of the dry weight of the ash, while the fixed solids contents of fly ashes from the LVH-EFW facility averaged more than 97%. This difference can also be explained by the differences inherent in the two incinerator designs. The high temperatures and highly oxidative conditions in the secondary chamber of the modular incinerator result in a much more efficient burnout of combustible material in the flue gas than that achieved in the RDF system. In addition, the semi-pyrolytic conditions in the primary chamber of the modular incinerator result in a very little carry-over of particulates in the flue gas stream, and hence, very little combustible material carried over to the air pollution control units. Conversely, the semi-suspension feed of the shredded refuse and the mixing of the air in the combustion chamber of the SWARU facility results in a high degree of particulate carry-over to the air pollution control devices. Subsequently, the greater mass of non-metallic material in the captured fly ash acts to dilute the metals which condense out of the flue gases and sorb onto the particulates.

Leachability

The ashes were subjected to the Sequential Batch Extraction Procedure (SBEP) to determine variations in leachate pH and potential inorganic contaminant mobility in water over a wide range of liquid-to-solid ratios (20:1 to 100:1). It is important to note that these results indicate changes in the leachability of contaminants with increasing contact with water and are not indicative of field leachate quality.

The variations in the pH of the leachates from the ashes of both facilities are shown in Figure 3. The bottom ash and fly ash from SWARU, as well as the bottom ash and CT/FF ash from the LVH-EFW facility, were highly alkaline (pH greater than 10) throughout the test. The pH of the SWARU bottom ash leachates declined from almost 12.5 to 10.5 over the five cycles. The pH of the SWARU fly ash leachates decreased from 12.2 to 11.1. The pH of the bottom ash and CT/FF ash leachates from the LVH-EFW facility remained fairly constant throughout the SBEP at about 10.5 to 11.5. Conversely, the boiler and economizer ashes from the LVH-EFW facility produced leachates with pH values which respectively increased from 6.5 and 7.5 in the first cycle of the test, to 7.6 and 8.6 in cycle 5.

The alkaline nature of most MSW incinerator residues, especially those associated with lime-based air pollution control systems, is well documented (e.g., US EPA, 1987; Brunner and Baccini, 1987; Sawell et al., 1987; Hjelmar, 1987). However, it appears that the lower pH of boiler and economizer ashes from modular incinerators is characteristic of this type of incinerator and may be related to the high degree of "fouling" which occurs in the heat recovery units of these facilities. The complex chemical and physical processes which result in the formation of these "fouling" deposits are not fully understood and require further study (Glen et al., 1988).

The cumulative total dissolved solids (TDS) in the leachates are given in Figure 4. In general, the bottom ashes from both facilities were much less soluble (an average of about 5%) than any of the fly ashes (an average of almost 35%), which is consistent with results from previous NITEP studies (Sawell et al., 1988; Sawell and Constable, 1988). The higher solubility is due in part to the smaller particle size distribution of the fly ash, and the presence of readily soluble components consisting of lime (if from a lime-based air pollution control system) and condensation products (salts) from the flue gas which sorb onto particles entrained in the flue gas stream.

The results also indicate that there is a substantial difference in the solubility of the fly ashes from the two facilities. Economizer ash was the least soluble fly ash (35%) from the LVH-EFW facility and it was still almost 3 times more soluble than the fly ash from SWARU. This was due to the modular design of the LVH-EFW system which

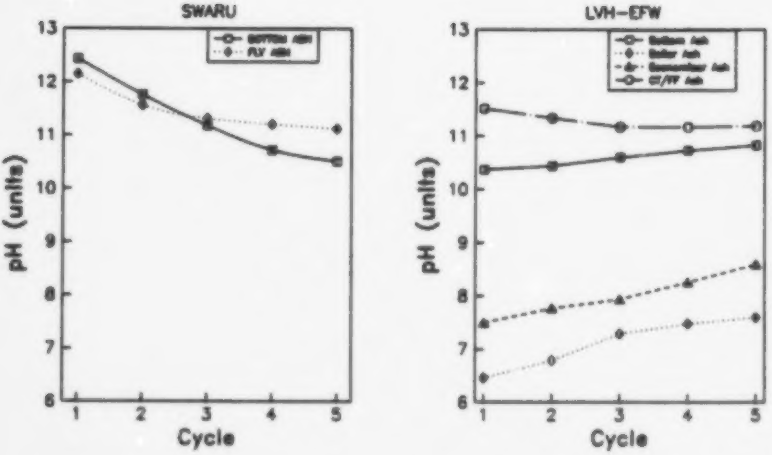

Figure 3. pH Readings from the Sequential Batch Extraction Procedure

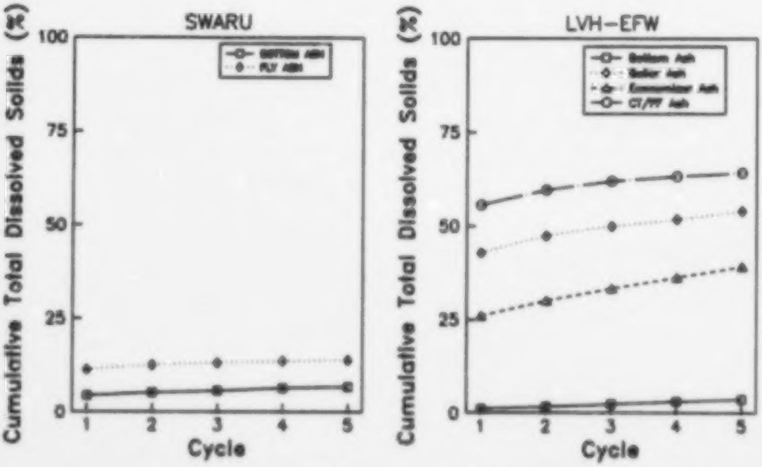

Figure 4. Cumulative Total Dissolved Solids Results from the Sequential Batch Extraction Procedure

results in less particulate carry-over from the primary and secondary chambers, and hence, a higher ratio of soluble flue gas condensation products to insoluble particulates than in the RDF system.

Fraction leached results for cadmium and zinc from the first cycle of the SBEP on ashes from both facilities are plotted against the pH of the corresponding leachate in Figure 5. Fraction leached is defined as the ratio of the mass of a contaminant released from the ash to the total mass of that contaminant originally present in the ash, expressed as a percentage. The results verify previous findings (e.g., Brunner and Baccini, 1987; Sawell et al., 1988) in that the solubility of Cd and Zn is affected by the pH of the leaching environment. In this case, the pH of the leaching environment was determined by the pH of the ash. Cadmium, which is not amphoteric, and zinc, which is mildly amphoteric, were readily soluble under mildly acid to neutral pH conditions but not under alkaline conditions.

The effect of pH on the solubility of lead is illustrated in Figure 6 where the fractions of lead leached in each of the five cycles of the SBEP are plotted against pH. Lead is considered strongly amphoteric and was more soluble under highly alkaline conditions than under mildly acid to neutral conditions where only very small fractions (less than 0.25%) were solubilized.

The results from the Sequential Chemical Extraction (SCE) Procedure are shown in Figure 7, and were used to estimate the potential leachability of the three metals in the ashes over the life of a landfill. The SCE is a step-wise separation of the total concentration of each metal into 5 distinct fractions, using progressively more aggressive digestion methods on the ash. Table 5 summarizes the descriptions and interpretations associated with each fraction. It is important to note that the interpretations are assumed generalities and that the results are not indicative of field leachate quality under the conditions suggested.

The results indicate that almost all of the cadmium present in all the ashes is potentially available for leaching upon contact with water or under the acidic conditions which may prevail in a co-disposal landfill (Fractions A + B). Over 58% of the lead present in the SWARU ashes and more than 78% of the lead present in the LVH-EFW ashes are potentially available for leaching over the life of a landfill (Fractions A + B + C). More than 78% of the zinc present in all ashes is potentially available for leaching over the life of a landfill.

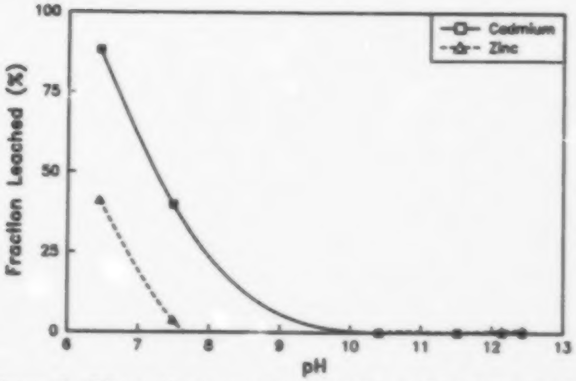

Figure 5. Effect of pH on the Solubility of Cadmium and Zinc

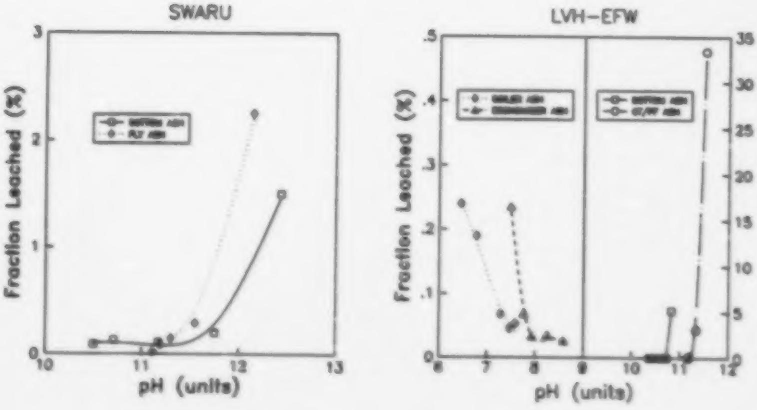

Figure 6. Effect of pH on the Solubility of Lead

Table 5. Summary of the Sequential Chemical Extraction Procedure Description and Interpretations

Fraction	Description	Interpretation
A	Ion exchangeable	Immediately available for leaching
B	Surface oxide and carbonate bound ions	Potentially available for leaching under acidic conditions
C	Iron and manganese bound metal ions	Potentially available for leaching under reducing conditions
D	Sulphide and organic matter bound ions	Unavailable for leaching under normal leaching conditions
E	Residual metal ions	Unavailable for leaching

The metal species profiles (i.e., the distribution over the five fractions) are considerably different for the LVH-EFW bottom ash and fly ashes. In general, larger fractions of the metals in the fly ashes are more easily leached than those in the bottom ash. This trend is not as evident in the SWARU ashes. The species profiles for lead in the SWARU ashes indicates that a larger fraction is more readily leachable from the bottom ash than from the fly ash. However, the profiles for cadmium and zinc in both SWARU ashes are very similar.

The reasons for the differences between SWARU and LVH-EFW ashes are probably related to differences in the combustion processes. For example, in the LVH-EFW system, the proportion of bottom ash to fly ash is generally 9:1. The metals in the bottom ash from the primary chamber are only exposed to semi-pyrolytic combustion conditions, whereas those in the fly ashes are also exposed to high temperatures and highly oxidative conditions. The subsequent speciation of the condensation products which sorb out of the flue gas stream onto well combusted particles is substantially different than the metal speciation in the bottom ash. Conversely, in the RDF system, shredded refuse is "blown" into the top of the incineration chamber resulting in a more

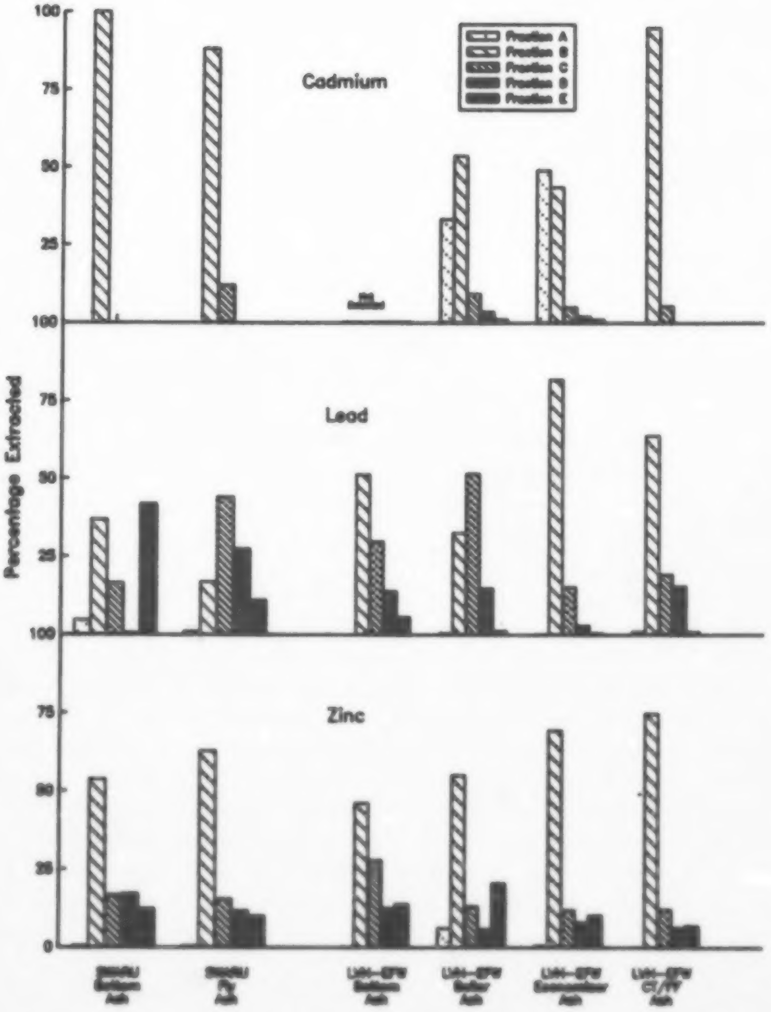

Figure 7. Results from the Sequential Chemical Extraction Procedure

even distribution of ash between bottom and fly ash. The proportion of bottom ash to fly ash generally ranges from 1:1 to 1.5:1, and metals in the bottom ash and fly ash are exposed to similar combustion conditions. As a result, the speciation of metals in the bottom ash and fly ashes from this RDF facility are similar.

Implications for Disposal Options

In October 1988, the MSW Incineration Sub-committee of the Canadian Council of Resource and Environment Ministers (CCREM) recommended management options for the disposal of MSW incinerator ash, as a part of the Canadian Operating and Emission Guidelines for MSW Incinerators (CCREM, 1988). One of the recommendations was that bottom ash be collected, stored and transported separately from the fly ash residue stream to facilitate treatment of the fly ash prior to disposal. The recommendation was based on the technical data generated through NITEP, which indicated that the chemical characteristics of bottom ash and fly ash from mass burn and modular municipal waste incinerators were significantly different, and that the potential for inorganic contaminant mobility from fly ash was high. Although the results from the LVH-EFW study provide further support for this recommendation, the results from the SWARU study indicate that fly ash from this RDF facility contains metal concentrations an order of magnitude lower than the LVH-EFW fly ashes. In addition, the RDF bottom ash and fly ash have similar chemical characteristics. Consequently, fly ash from this RDF system may not need treatment prior to disposal, and hence, may be suitable for landfill disposal. Further study is currently being conducted by Environment Canada and the U.S. Environmental Protection Agency to obtain a better understanding of the behaviour of metals in RDF systems.

Another CCREM guideline recommendation involved disposal options for the different types of residues. It was recommended that the chemical characteristics of bottom ash should permit its disposal in a municipal landfill. Although the results from the SBEP indicate that bottom ash is fairly insoluble in water, it is important to note that metal solubility from bottom ash is highly pH dependent. Since the solubility of metals increases under acidic conditions, (as illustrated by the SCE analyses) and the bio-degradation of MSW would hasten a drop in pH of the leaching environment, it would be appropriate to segregate the MSW incinerator ash within a landfill to maintain a more stable pH environment and reduce potential metal solubility.

CONCLUSIONS AND RECOMMENDATIONS

The chemical properties of the ashes from the two facilities were dependent on the type of incinerator design used. For example, ferrous metal removal in the pre-processing of refuse at the RDF facility reduced concentrations of cadmium, lead and zinc in the fly ash. The RDF system also appeared to distribute the volatile metals evenly between bottom ash and fly ash, whereas much higher concentrations of volatile metals were measured in the fly ashes than bottom ash from the modular incinerator.

Contaminant leachability from MSW incinerator ashes is dependent upon several factors, including the pH of the leaching environment, the combustion conditions and the speciation of the metals. Bottom ashes from both facilities were found to be relatively insoluble in water in comparison to the fly ashes. They may be considered suitable for landfill disposal but should be placed in an area where the pH will remain alkaline (i.e., bottom ashes should not be mixed with MSW).

The results of this and other studies indicate that appropriate disposal or utilization options for MSW incinerator ashes should be based on the incineration technology used. The chemical characteristics of bottom ash and fly ash from a refuse derived fuel system with ferrous recovery were similar and separate collection to facilitate treatment of fly ash may not be necessary. Conversely, the characteristics of ashes from a modular incinerator were very different, and separate collection of bottom ash and fly ash, and treatment of the fly ash, is warranted.

ACKNOWLEDGEMENT

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The Food Chain as a Source of Human Exposure from Municipal Waste Combustion

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The Food Chain as a Source of Human Exposure from Municipal Waste Combustion

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ABSTRACT

The food chain is the primary pathway of human exposure for a large class of lipophilic compounds, such as dioxins, DDT, and other pesticides. Since municipal waste combustors release both metals and organics into the environment, the food chain pathway must be considered as a potential source of human exposure. This paper presents estimates of human exposure through the food chain for a typical municipal waste combustor. A Monte Carlo uncertainty analysis is performed to characterize variability in exposure estimates.

INTRODUCTION

Ninety-nine municipal waste combustors (MWCs) are currently operating in the United States, having the incineration capability of more than 47,000 tons of waste. By the year 2000, that capacity is estimated to reach 250,000 tons (1). As the use of municipal waste incineration as a waste management alternative has increased, public concern over possible environmental and human health effects has also increased. Of particular concern are potential exposures through the food chain.

The food chain is the primary pathway of human exposure for a large class of organics, such as dioxin, PCBs, PCPs, DDT, and other pesticides (2-4). Because many pollutants emitted by MWCs are lipophilic, extremely persistent compounds, they tend to sorb strongly to air particles, soil, and sediment and to bioaccumulate in living organisms. As a result, the food chain can be a major pathway of exposure to pollutants emitted by MWCs (5). It is the purpose of this paper to assess the magnitude of human exposure through the food chain for two pollutants released by MWCs: cadmium and 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD, commonly referred to as dioxin). These pollutants were chosen as representatives of two chemical classes: metals and organics. Cadmium is a metal which presumably will enter the food chain primarily through vegetative root uptake, while dioxin is an extremely lipophilic compound which will bioconcentrate in beef and milk (2).

When models are used as the basis for making estimates of exposure in risk assessment, the question arises as to the sensitivity of model predictions to uncertainties in model input parameters (6). A first step in the direction of answering this question is the identification of model parameters which make the greatest contribution to overall model error. These represent the parameters which must be determined most accurately in experiments. The second step is to determine probability distributions for the most sensitive parameters and propagate these distributions through the model to obtain a characterization of uncertainty in model output. Both of these steps will be attempted for cadmium and dioxin.

THE TERRESTRIAL FOOD CHAIN MODEL

The Terrestrial Food Chain (TFC) Model, a multimedia transport model developed at the Oak Ridge National Laboratory, was used to study the accumulation of cadmium and dioxin in the food chain. The TFC Model uses atmospheric deposition rates to calculate the human daily pollutant intake through the food chain. Deposition rates and atmospheric concentration values for this study were taken from estimated values for a proposed mass burn incinerator in Tampa, Florida, and are listed in Table 1.

Table 1. Deposition Rates and Concentration Values for Tampa

	Deposition ($g/m^2/yr$)	Concentration (ug/m^3)
Cadmium	5.298E-3	1.944E-3
Dioxin	2.350E-7	1.156E-7

The methodology used to estimate the uptake of chemicals by vegetation and forage crops is given by the following general equation:

$$C_p = Dy * (A_d + U_s + T_s) \quad (1),$$

where C_p = concentration in plants (ug/kg);
Dy = annual deposition rate ($g/m^2/yr$);
A_d = atmospheric deposition component (ug/g DW);

U_s - uptake from soil component (ug/g DW);
T_a - air-to-plant transfer component (ug/g DW).

This equation simply states that the concentration of a pollutant in vegetation results from one of or all of the following three methods (7): (1) an atmospheric deposition component in which the contaminant is deposited directly onto the plant; (2) a soil uptake component in which the pollutant is first deposited on the soil and then taken up by the roots of the plant and translocated to the edible portion of the vegetation; (3) an air-to-plant transfer component in which the pollutant is absorbed by the plant from the surrounding contaminated air.

The atmospheric deposition component is calculated by:

$$A_d = \frac{K2 * D_y * R_p * [1.0 - \exp(-K_p * T_p)]}{Y_p * K_p} \qquad (2).$$

where K2 - units conversion factor of 1000;
D_y - annual deposition rate of pollutant (g/m^2/year);
R_p - interception fraction of the edible portion of the plant (unitless);
K_p - plant surface loss coefficient (1/years);
T_p - time period of exposure to deposition per harvest of the edible portion of the plant (years);
Y_p - yield or standing crop biomass of the edible portion of the plant (kg DW/m^2).

Vegetative uptake from soil is given by:

$$U_s = \frac{B_s * (K1 * D_y * [1.0 - \exp(-K_s * T_c)])}{Z * Bd * K_s} \qquad (3).$$

where B_s - soil-plant bioconcentration factor (ug/g DW)/(ug/g soil);
K1 - units conversion factor of 100;
K_s - soil loss constant (1/years);
T_c - period of long-term buildup in soil (years);
Z - soil depth (cm);
Bd - Soil bulk density (g/cm^3).

The soil loss constant is the sum of all soil losses due to leaching, (K_l), or degradation and volatilization, (K_{dv}), and therefore:

$$K_s = K_l + K_{dv} \qquad (4).$$

The air-to-plant transfer component is given by:

$$T_a = \frac{B_v * (C_v + C_p)}{Ba} \qquad (5).$$

where B_v = air-to-vegetation bioconcentration factor (ug/g DW)/(ug/g air);
C_v = concentration of pollutant in air due to volatilization from soil (ug/m^3 air);
C_y = average atmospheric concentration of pollutant due to direct emission from a municipal waste incinerator (ug/m^3 air);
Da = density of air (1190 g/m^3 at 25° C).

The estimation of the concentration of the pollutants in food and forage crops is just the initial phase in calculating total exposure from the food chain. The concentrations in animal tissues and milk that result from animals ingesting contaminated forage and grains must also be determined. Pollutant concentration for each animal tissue group is modeled via:

$$A_i = (Qf_i * C_f + Qg_i * C_g + Qs_i * Sc) * B_a \qquad (6),$$

where A_i = concentration of pollutant in the animal tissue (ug/g DW);
Qf_i = quantity of forage eaten by the animal each day (kg DW/day);
Qg_i = quantity of locally grown grain eaten by the animal each day (kg DW/day);
Qs_i = quantity of soil eaten by the animal each day (kg soil/day);
C_f = concentration of pollutant in animal forage (ug/g DW);
C_g = concentration of pollutant in animal grain feed (ug/g DW);
Sc = soil concentration of pollutant after the total time period of deposition (ug/g soil);
B_a = biotransfer factor for the animal tissue group (ug day/kg DW).

The total exposure to a pollutant is computed by summing the pollutant intakes from each type of vegetation and from each animal tissue group.

UNCERTAINTY ANALYSIS

The first step in the uncertainty analysis of the TFC Model was to determine those model parameters which make the greatest contribution to overall model output variability. Therefore, a sensitivity analysis was performed using a Monte Carlo sensitivity and uncertainty analysis code called PRISM (8), which employs a Latin Hypercube sampling method. For the sensitivity analysis, each parameter is assumed to have a normal distribution with the standard deviation set at 1% of the mean (8). Monte Carlo sampling is then performed to determine the parameters having the greatest impact on model output variability. Since all parameters are assumed to have the same probability distribution in a sensitivity analysis, those contributing most to model output variability are the most sensitive parameters.

Parameters that contribute more than 2% to total model output variability are listed in Table 2 for cadmium and Table 3 for dioxin. Thirteen variables passed this criteria for cadmium, and they account for almost 60% of the variation in total daily intake of cadmium. Eight parameters passed this criteria for dioxin and contribute over 60% of the variation in the total daily intake of dioxin. The individual contributions of each parameter to this variation are summarized below:

Table 4. % of Parameter Influence on Total Daily Intake

Cadmium		Dioxin	
Precip	17.11%	Ba.dairy	22.06%
Evap	6.64	Ca.dairy	9.88
Irigat	4.41	Ba.beef	8.66
Cp.gfr	4.13	Qs.beef	6.49
Kd	3.70	Bd	5.33
Tc	3.67	Qs.dairy	5.00
Theta	3.23	Fa.dairy	3.33
Rp.legd	2.90	Bg	2.39
Qf.dairy	2.60		
Fa.pork	2.33	Total	63.14%
Ca.pork	2.33		
Br.pot	2.20		
Ba.pltry	2.12		
Total	57.37%		

The major contributors to the variance of cadmium intake levels are precipitation, evapotranspiration, and irrigation. These parameters are important in determining equilibrium soil concentrations of cadmium. Since ingestion of contaminated vegetation is the main route of exposure to cadmium and root uptake is responsible for almost 82% of the exposure of this pathway (see RESULTS), these variables are the prime contributors to variation in cadmium intake.

The primary sources of variation in dioxin intake are the biotransfer factor and the daily human consumption for dairy products. A vital characteristic of dioxin is that due to its high lipophilicity, it readily accumulates in living organisms. Travis and Hattemer-Frey have estimated that meat and dairy products account for 77% of human exposure to dioxin (2). Therefore, the biotransfer factor for dairy products and the daily human consumption of dairy products are parameters which contribute greatly to dioxin intake variability.

The second step in the uncertainty analysis of the TFC Model was to determine the actual probability distributions for the most sensitive parameters and to propagate these distributions through the model to obtain a characterization of uncertainty in model output. A literature search was therefore performed to determine the proper probability distribution for each parameter. The PRISM code allows for the input of several types of distributions. However, the code is designed to take only normal distributions as input and to convert those numbers to the type of distribution specified by the user. Parameter values obtained from the literature were plotted on probit paper to determine if the parameter adhered to a lognormal distribution. All parameters were found to be lognormally distributed, except *theta* which was assumed to be normally distributed. Table 5 summarizes the distributions that were used as input:

Table 5. Probability Distributions of Most Influential Parameters

Parameter	Arithmetic Mean	Standard Deviation	Distribution Type
For cadmium:			
Ba.pltry	0.0818	0.0784	lognormal
Br.pot	0.03	0.0003	lognormal
Ca.pork	11.13	2.98	lognormal
Cp.gfr	5.21	0.59	lognormal
Evap	64.81	17.98	lognormal
Fa.pork	0.4499	0.0414	lognormal
Irigat	32.99	27.08	lognormal
Kd	9.19	7.19	lognormal
Precip	88.18	34.74	lognormal
Qf.dairy	7.315	4.251	lognormal
Rp.legd	0.0031	0.001	lognormal
Tc	65.0	49.50	lognormal
Theta	0.22	0.07	normal
For dioxin:			
Ba.dairy	0.6051	0.8413	lognormal
Ba.beef	0.6275	0.8096	lognormal
Bd	1.40	0.11	lognormal
Bg	0.7978	1.37	lognormal
Ca.dairy	30.23	13.58	lognormal
Fa.dairy	0.3738	0.1048	lognormal
Qs.beef	0.7559	0.7674	lognormal
Qs.dairy	0.7559	0.7674	lognormal

RESULTS

Table 6 shows that cadmium intake due to emissions from MWCs is due mainly to the consumption of contaminated vegetation (51.3%). Almost 42% of total daily intake is attributed to the root uptake component, while atmospheric deposition contributes only 9.4%.

Table 6. Breakdown of Exposure Pathways for Cadmium
(% contributions to average daily intake)

1. Direct ingestion of contaminated soil :	6.9%
2. Consumption of contaminated vegetation :	
Atmospheric deposition component	9.4%
Root uptake component	41.9%
Air-to-plant transfer component	0.0%
3. Consumption of contaminated meat and dairy products:	
Soil-Animal-Human Route	16.3%
Plant-Animal-Human Route	25.5%

The analysis also determined the breakdown of cadmium daily intake according to general food groups. Garden fruits and beef are the primary plant tissue group and animal tissue group, respectively, contributing to total daily intake. Since contaminated vegetation is the most prominent exposure pathway for cadmium, the divisions for vegetation are listed below with their corresponding estimates of percentage influence:

Table 7. Contribution to Daily Intake of Cadmium by Vegetation Food Groups

Source	Percentage of the Total Intake
Garden fruits	19.1%
Root vegetables	9.8%
Leafy vegetables	9.5%
Fresh legumes	8.7%
Potatoes	2.3%
Dried legumes	1.9%

The consumption of contaminated meat and milk products contributes 41.75% of total human exposure to cadmium through the food chain. This particular exposure route can be further subdivided into soil-animal-human and plant-animal-human. The latter pathway is the most prolific, contributing more than 25% of total cadmium intake.

A final goal of the uncertainty analysis was to determine a probability distribution for the daily human intake of cadmium due to emissions of the contaminant from MWCs (in this case, based on deposition rates for a proposed incinerator in Tampa). Figure 1 shows that these values fit a lognormal distribution, and their statistical summary is as follows:

Table 8. Distribution of Daily Cadmium Intake Values (ug/day)

Geometric Mean	Geo St Dev	Minimum	Maximum
0.99	0.15	0.55	2.80

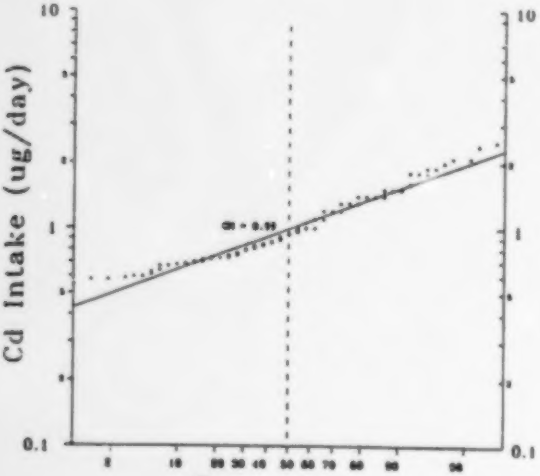

Figure 1. Cadmium Daily Intake Values

Table 9 shows that the primary route of human exposure to dioxin from the food chain is through meat and dairy products, contributing 99% of total exposure:

Table 9. Breakdown of Exposure Pathways for Dioxin (% contributions to average daily intake)

1. Direct ingestion of contaminated soil :	<.01%
2. Consumption of contaminated vegetation :	
Atmospheric deposition component	0.5%
Root uptake component	<.1%
Air-to-plant transfer component	0.3%
3. Consumption of contaminated meat and dairy products:	
Soil-Animal-Human Route	44.1%
Plant-Animal-Human Route	54.9%

The soil-animal-human route accounts for 44.1% of this exposure pathway, and the plant-animal-human mechanism composes 54.9%. Dairy products and beef dominate as the primary contributing animal tissue groups, while no plant tissue group exhibits more than 1% influence. The breakdown of contributions according to the general food groups by meat is summarized by the following Table 10:

Table 10. Contribution to Daily Intake of Dioxin by Meat Food Groups

Source	Percentage of the Total Intake
Dairy	52.8%
Beef	42.5%
Beef liver	3.6%
Lamb	0.1%
Pork	<.003%
Eggs	<.001%
Poultry	<.001%

The uncertainty analysis also determined a probability distribution for the daily human intake of dioxin due to MWC emissions. Figure 2 shows that dioxin's intake values adhere to a lognormal distribution, and their statistical description is as follows:

Table 11. Distribution of Daily Dioxin Intake Values (ug/day)

Geometric Mean	Geo St Dev	Minimum	Maximum
0.0063	0.0850	0.0038	0.0091

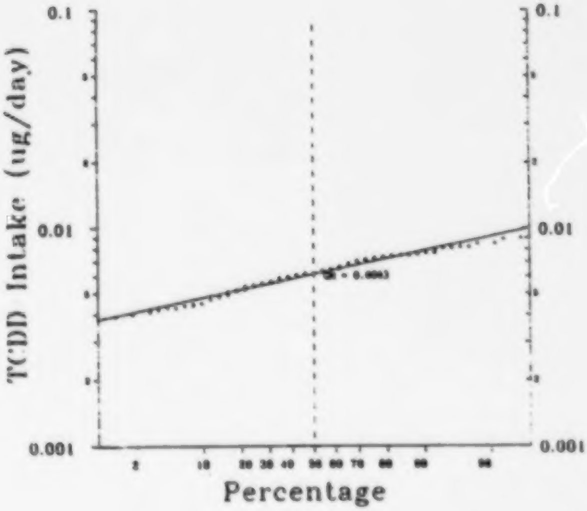

Figure 2. Dioxin Daily Intake Values

Figures 3 and 4 show the 95% confidence intervals for the daily intakes of cadmium and dioxin, respectively, from the various exposure pathways. An inhalation estimate is included to demonstrate the magnitude of the food chain's contribution to intake. Only 4.3% of the total average individual intake (inhalation plus food chain) is attributable to inhalation for cadmium, while less than 1% is due to inhalation for dioxin. Also, the 95% confidence interval for total intake is less than a factor of 1.6 times the best estimate for dioxin exposure and is less than a factor of 2.1 times the best estimate for cadmium exposure. Thus, this study confirms previous studies that have shown that the food chain can be an important contributor to human exposure of organic chemicals released from MWCs (2,3).

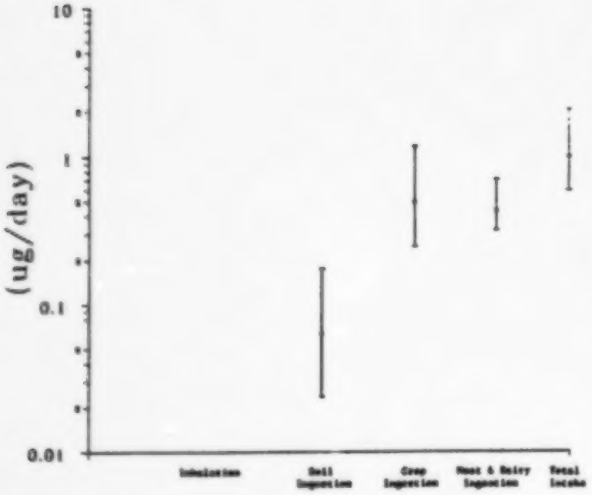

Figure 3. Daily Intake of Cadmium from Various Exposure Pathways. Error Bars Represent 95% Confidence Intervals.

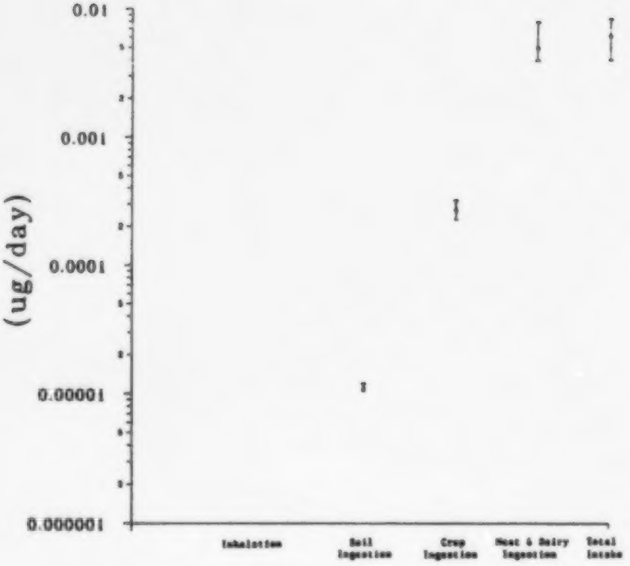

Figure 4. Daily Intake of Dioxin from Various Exposure Pathways. Error Bars Represent 95% Confidence Intervals

CONCLUSION

The food chain can be a primary pathway of human exposure to contaminants released by MWCs. An uncertainty analysis was performed on the TFC Model to determine the extent of this exposure for cadmium and dioxin. Two main pathways of exposure were analyzed: consumption of contaminated vegetation and consumption of contaminated meat and dairy products. Three routes of food and forage crop contamination were considered: (1) atmospheric deposition; (2) uptake from the soil; and (3) air-to-plant transfer. The analysis showed that cadmium intake is due mainly to the consumption of contaminated plants, especially garden fruits. The main pathway of vegetation contamination is via root uptake. Dioxin intake occurs primarily through the ingestion of contaminated meat and dairy products, especially milk and beef.

Based on deposition rates for a proposed MWC in Tampa, the distributions for cadmium and dioxin daily intake values were determined. Cadmium values ranged from 0.55 to 2.80 (ug/day), having a geometric mean of 0.99. Also, the 95% confidence interval was less than a factor of 2.1 times the best estimate for cadmium exposure. Dioxin values varied from 0.0038 to 0.0091 (ug/day) and had a geometric mean of 0.0063, with the 95% confidence interval less than a factor of 1.6 times the best estimate for dioxin total intake. Inhalation was shown to be a minor source of cadmium and dioxin intake in comparison to the contributions of the food chain pathways.

The work described in this paper was funded by the U.S. Environmental Protection Agency, but this paper has not been reviewed in accordance with the Agency's administrative review policies. Therefore, the contents of this paper do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

Table 2. Cadmium Parameters Exhibiting Greatest Influence on Total Daily Intake

Ba.pltry : Biotransfer factor for poultry representing the relationship between equilibrium concentration of pollutant in an animal tissue and daily intake of pollutant (ug pollutant day/kg animal tissue DW).

Br.potato: Soil-to-plant bioconcentration factor for potatoes consumed by humans (ug pollutant/g plant tissue DW) / (ug pollutant/g soil).

Ca.pork : Daily human consumption of pork (g animal tissue DW/day).

Cp.gfr : Daily dietary consumption, by age and sex, of garden fruits (g plant tissue DW/day).

Evap : Average annual evapotranspiration (cm/year).

Fa.pork : Fraction of pork consumed by humans that is locally produced (unitless).

Irigat : Average annual irrigation (cm/year).

Kd : Soil-to-water partitioning coefficient (ml/g).

Precip : Average annual precipitation (cm/year).

Qf.dairy : Quantity of forage eaten by an animal per day (kg/day).

Rp.legdr : Interception fraction for the edible portion of dried legumes consumed by humans (unitless).

Tc : Total time period over which deposition occurs (years).

Theta : Soil volumetric water content (ml/cm**3).

Table 3. Dioxin Parameters Exhibiting Greatest Influence on Total Daily Intake

Ba.beef : Biotransfer factor for beef representing the relationship between equilibrium concentration of pollutant in an animal tissue and daily intake of pollutant (ug pollutant day/kg animal tissue DW).

Ba.dairy: Biotransfer factor for dairy representing the relationship between equilibrium concentration of pollutant in an animal tissue and daily intake of pollutant (ug pollutant day/kg animal tissue DW).

Bd : Soil bulk density (g/cm**3).

Rg : Soil-to-plant bioconcentration factor for animal grain feed (ug pollutant/g plant tissue DW) / (ug pollutant/g soil).

Ca.dairy: Daily human consumption of the dairy group (g animal tissue DW/day).

Fa.dairy: Fraction of the dairy group consumed by humans that is locally produced (unitless).

Qs.beef : Quantity of soil eaten by an animal per day (kg/day).

Qs.dairy: Quantity of soil eaten by an animal per day (kg/day).

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A PERSPECTIVE ON HUMAN EXPOSURE TO PCDDs AND PCDFs EMITTED FROM MUNICIPAL WASTE COMBUSTORS

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A PERSPECTIVE ON HUMAN EXPOSURE TO PCDDs AND PCDFs EMITTED FROM MUNICIPAL WASTE COMBUSTORS

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ABSTRACT

This paper discusses the following key issues concerning human exposure to polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) emitted from modern municipal waste combustors (MWCs): (1) Are MWCs the major source of PCDD/PCDF input into the environment? (2) Are environmental concentrations around MWCs substantially elevated relative to background levels? and (3) Are MWCs the major source of human exposure to PCDDs and PCDFs? Current scientific evidence indicates that: (1) combustion sources *in general* (including steel mills, copper smelting plants, motor vehicles, hospital incinerators, and MWCs) are major sources of PCDD/PCDF input in the environment; (2) environmental concentrations of PCDDs and PCDFs around operating MWCs are not substantially elevated; and (3) 99% of human exposure to PCDDs and PCDFs is from background environmental contamination, even for individuals living near a modern municipal waste combustor.

INTRODUCTION

Polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs), including the highly toxic 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD, commonly referred to as "dioxin"), are the subject of much public concern and debate. Because TCDD is the most potent chemical carcinogen ever evaluated by the Environmental Protection Agency (1), there is s[eculation that exposure to even small amounts of dioxin could lead to serious adverse human health effects. As the controversy surrounding municipal waste combustion intensifies, government agencies, public officials, and the public, question if municipal waste combustion is a feasible waste management alternative. Because it has become a formidable challenge for any one individual to critically evaluate the myriad of information available, the majority of the general public and segments of the scientific community seem to have a limited knowledge of some key issues which are critical to but often overlooked in this debate.

This paper is not intended to resolve the incineration debate but to examine the following three issues concerning human exposure to facility-emitted dioxins and furans: (a) Are municipal waste combustors (MWCs) the major source of PCDD/PCDF input into the environment? (b) Are environmental concentrations of PCDDs/PCDFs around MWCs substantially elevated? and (c) Are MWCs the major source of human exposure to PCDDs and PCDFs?

We acknowledge that many people are quite concerned about dioxin emitted from MWCs and that the existing data concerning environmental concentrations of dioxin are not sufficient to make any definitive statements. It is not our position that MWCs are safe, nor are we recommending municipal waste combustion as the optimum solution to waste disposal problems. Many other issues (such as health risks associated with ash management and the economic aspects of incineration) are not addressed in this paper and require further investigation. Our analysis, however, indicates that: (a) combustion sources in general (not just MWCs) are the major source of environmental dioxin; (b) background levels of dioxin are alarmingly high; and (c) dioxin levels near MWCs are not substantially elevated relative to background levels. Thus, individuals expressing concern over dioxin emissions from MWCs need to produce data and analyses showing why that concern is warranted.

ISSUE 1 - ARE MWCs THE MAJOR SOURCE OF PCDD/PCDF INPUT INTO THE ENVIRONMENT?

PCDDs and PCDFs are formed during many combustion processes (2). Principal environmental sources of PCDDs and PCDFs are suspected to be: (a) industrial sources, including metal processing and treatment plants, pulp and paper mills, steel mills, and copper smelting plants (3-6); (b) motor vehicle emissions (7-9); (c) chemical manufacturing processes (10); (d) combustors, including chemical, industrial, hospital, and municipal waste combustors (10-12); and (e) forest fires and residential wood burning (12,13).

INDUSTRIAL SOURCES

Marklund *et al.* (3) and Rappe *et al.* (4) reported that PCDD/PCDF emissions from high-temperature industrial processing plants, such as steel mills and copper smelting plants, seem to be of the same magnitude as emissions from MWCs. The Swedish EPA (14) estimated that emissions of PCDDs and PCDFs from these types of plants in Sweden range from 50 to 150 grams of Toxic Equivalents Dioxins and Furans (TEDFs) per year, while normal-sized, modern MWCs (50 to 200,000 tons of waste per year capacity) emit about 1 to 100 grams TEDFs per year (15,16). [The toxicity of different PCDD and PCDF isomers is typically expressed in terms of TEDFs, which relates the toxicity of all PCDD/PCDF compounds to the known toxicity of TCDD using one of several weighing schemes (17)]. Because there are many more industrial sources, however, their contribution to total PCDD/PCDF input could be much greater than the contribution from MWCs (4). Thus, various researchers argue that industrial sources may contribute equal or even larger amounts of PCDDs and PCDFs into the environment than MWCs.

MOTOR VEHCILES

Studies have shown that motor vehicle emissions may be a larger source of PCDD/PCDF input than MWCs. Ballschmiter *et al.* (7) argued that a more prevalent source of PCDD/PCDF input than municipal waste combustion must exist to account for the widespread background contamination of PCDDs and PCDFs and suggested that the ubiquitous, non-point character of motor vehicle emissions "strongly recommends this source for consideration as a major environmental input."

Marklund *et al.* (8) found that total emissions of PCDDs and PCDFs from cars in Sweden using unleaded gasoline were 10 to 100 grams TEDFs (Eadon method) per year, which is equivalent to the amount of PCDDs and PCDFs emitted from two to 20 MWCs of normal size and technology. Thus, in Sweden, motor vehicles and MWCs emit about the same amount of PCDDs and PCDFs. Jones (18) reached the same conclusion for the United States and contended that since motor vehicle emissions are so widespread, roadside exposures are equal to or greater than exposures to PCDDs and PCDFs emitted from MWCs.

Rappe *et al.* (9) measured PCDD/PCDF levels in ambient air from (a) a traffic tunnel; (b) an industrial area 1500 meters downwind of a MWC; and (c) a suburban area 13 kilometers from Hamburg, West Germany. PCDD/PCDF levels in the traffic tunnel [28 picograms/cubic meter of air (pg/m^3)] were two times higher than levels downwind from the MWC (13 pg/m^3) and nine times higher than the suburban air concentration (3.0 pg/m^3) (9). Rappe et al. (9) concluded that "measurements made in the traffic tunnel clearly indicate that motor vehicles are a source of PCDDs and PCDFs in the ambient air." Hence, several researchers confirm that motor vehicle emissions may also be a significant source of PCDD/PCDF input into the environment. It should be noted, however, that these two studies evaluated emissions from European motor vehicles and apply only to vehicles run on leaded gasoline. The contribution of vehicles run on unleaded gasoline and diesel fuel has yet to be determined.

PULP AND PAPER MILLS

Pulp and paper mills that use chlorine bleaching processes may be another source of PCDD/PCDF input (19). Tests conducted at mills using a chlorine bleaching process showed that: (a) TCDD is formed in trace amounts during the bleaching of hard- and softwood pulps with chlorine and chlorine derivatives, and (b) the relative amounts of TCDD found in mill discharges varied dramatically from mill to mill (20). Swanson *et al.* (6) analyzed the effluents from mills in Sweden that use such processes and found that PCDD/PCDF releases ranged from 2 to 5.8 g TEDFs (Eadon) per year. They estimated that all pulp and paper mills in Sweden as a whole discharge 5 to 15 g TEDFs per year. Their *preliminary* findings suggest that the amount of PCDDs and PCDFs emitted from pulp and paper mills in Sweden is small relative to other known sources.

OTHER SOURCES

Sheffield (13) estimated that forest fires in Canada release 58,700 grams of dioxins per year, while residential wood burning may input an additional 1,800 g/year and concluded that "forest fires are estimated to be the largest source of PCDD release to the environment." Bumbaco (21) quantified PCDDs and PCDFs emitted from a hospital incinerator in Victoria, British Colombia. Average PCDD/PCDF emissions from this *single incinerator* was 23 g/yr, which suggests that hospital incinerators may be a dominant source of environmental PCDDs and PCDFs.

SUMMARY

These data indicate that combustion processes *in general* (not just MWCs) are candidate sources of PCDD and PCDF input into the environment and that it is premature to conclude that MWCs are *the* major source. The magnitude of PCDDs and PCDFs released from other combustion sources known to emit dioxins and furans (i.e., copper smelting plants, steels mills, motor vehicles, and hospital incinerators) needs to be quantified before definitive statements about the major source(s) of PCDD/PCDF input can be made.

ISSUE 2 - ARE PCDD/PCDF CONCENTRATIONS AROUND MWCs ELEVATED?

To determine if environmental concentrations of PCDDs and PCDFs around operating MWCs are truly elevated, they must be evaluated in the context of *background* environmental concentrations to which individuals are continuously exposed. Measured concentrations of PCDDs and PCDFs in air, soil, and animal samples obtained from sites near operating MWCs are scarce. The existing data, however, provide little evidence that PCDD/PCDF concentrations around these facilities are substantially elevated relative to existing background levels.

URBAN AIR

Table 1 shows that urban air concentrations are remarkably similar in all industrialized countries where measurements have been made (U.S., West Germany, Sweden, Japan, and New Zealand), ranging from 1.3 to 9.6 pg/m^3, with a geometric mean concentration of 3.4 pg/m^3.

TABLE 1. CONCENTRATION OF PCDDs AND PCDFs IN URBAN AIR

Location	Concentration (pg/m^3)	Reference
West Germany	3.0	9
West Germany	8.7	22
U.S. (Indiana)	2.2	23
U.S. (New York)	2.5	24
U.S. (California)	9.6	25
U.S. (Ohio)	4.5	26
Japan	8.7	27
Sweden	1.3	28
New Zealand	6.7	29
Geometric mean	4.3	

We could locate only one study, however, that reported both background urban air concentrations and the concentration of PCDDs and PCDFs in air near an operating MWC. Rappe *et al.* (9) found that measured concentrations of PCDDs and PCDFs in air near an operating MWC in West Germany (13.0 pg/m^3) were about four times higher than measured background air levels in West Germany (3.0 pg/m^3). That same study, however, found that PCDD and PCDF levels in air 100 meters from a motorway and 600 meters from a copper smelting plant (34.0 pg/m^3) were about 11 times higher than measured background air levels and three times higher than levels measured around the MWC (9). These data support the theory that numerous combustion sources (not just MWCs) are contributing to elevated levels of PCDDs and PCDFs measured in the ambient air of industrialized countries.

URBAN SOIL

PCDD/PCDF levels measured in background urban soil and in soil around operating MWCs are presented in Table 2.

TABLE 2. CONCENTRATION OF PCDDs AND PCDFs IN URBAN SOIL AND IN SOIL FROM SITES NEAR OPERATING MWCs.

Location	Levels in urban soil (ng/g)	Levels Near MWCs (ng/g)	Reference
United States	1.32	NA[a]	30
Japan	NA[a]	5.30	31,32
Canada	1.82	0.38	33
Italy	0.01	0.25	34
Denmark	0.29	0.21	35
Geometric mean	0.29	0.57	

[a] Measurement not taken

Yasuhara *et al*. (31) and Morita *et al*. (32) measured PCDD/PCDF levels in soil near four MWCs in Japan. (Incinerator type, age, and distance from the incinerator at which samples were taken were not specified). The geometric mean concentration of those samples (5.3 nanograms (ng)/g) is about 18 times higher than the mean world background concentration (0.29 ng/g) but only about three times greater than the US/Canada mean background urban concentration (1.6 ng/g). Since background concentrations of PCDDs and PCDFs in Japanese soil were not reported, it is impossible to determine if soil concentrations near these incinerators were elevated relative to background levels.

McLaughlin and Pearson (33) measured soil concentrations of PCDDs and PCDFs at 11 sites 0.1 to 2.1 kilometers (km) away from the SWARU MWC which has been operating in Hamilton, Ontario, Canada, since 1973, and at two urban sites 4.5 and 5.6 km away. The geometric mean concentration for the 11 sites near the MWC (0.38 ng/g) was about five times lower than the geometric mean concentration for the two urban sites (1.82 ng/g) but equivalent to the world background concentration (0.29 ng/g).

Berlincioni and di Domenico (34) assayed soil samples collected from five sites within a one km radius of a MWC which has been operating in Florence, Italy, since 1973. Background levels were determined at two sites eight km from the MWC. The geometric mean concentration near the MWC (0.25 ng/g) is 25 times higher than Italian background levels (0.01 ng/g) but virtually identical to mean world background levels (0.29 ng/g).

Finally, Nielsen and Lokke (35) examined soil samples from three "relatively unpolluted sites" 0.1 to 0.5 km from a small MWC and found that PCDD/PCDF concentrations near the MWC (0.21 ng/g) were not elevated relative to Danish or world background concentrations (0.29 ng/g). Depending on release point and plume characteristics, however, maximum soil concentrations may have occurred further away from the incinerator.

These data are not sufficient to make definitive statements about whether the concentration of PCDDs and PCDFs around operating MWCs are substantially elevated. For example, if the Japanese data are included in the geometric mean, soil levels near MWCs appear to be about two times higher than background concentrations. Conversely, if the Japanese data are not included in the geometric mean, PCDD/PCDF levels near MWCs are not elevated relative to background concentrations.

COW MILK

Although analysis of dioxin in commercial milk is limited, existing measurements indicate global contamination of dairy products. PCDD/PCDF levels in commercial milk samples (whole milk basis) from industrialized countries range from 0.6 to 3.1 pg/g, with a geometric mean concentration of 1.1 pg/g (Table 3). Rappe *et al.* (15) analyzed single milk samples from cows that grazed on farmland located: (a) 16 km from a small MWC and 20 km from a large MWC near Bern, Switzerland; (b) 0.2 to one km from a medium-sized MWC near Hunzenschwil, Switzerland; (c) 0.3 to one km from the same MWC in (b); and (d) one km from a production site for chlorinated products in Rheinfelden, West Germany. PCDD/PCDF concentrations for the three MWC samples ranged from 0.84 to 3.2 ng/kg, with a geometric mean concentration of 2.2 ng/kg, which is two times higher than world background concentrations. Mean PCDD/PCDF levels in milk taken from cows grazing near the production plant for chlorinated products (2.7 pg/g), however, were slightly higher than levels in the MWC samples (15), which demonstrates that other processes (not just MWC) are capable of contributing to elevated levels of PCDDs and PCDFs in cow milk.

TABLE 3. CONCENTRATION OF PCDDs AND PCDFs IN COMMERCIAL MILK SAMPLES*

Country	Concentration (pg/g)	Reference
Switzerland	0.56	15
West Germany	0.81	36
Japan	0.99	37
Canada	3.10	38
Geometric mean	1.1	

* Whole milk basis

ISSUE 3 - ARE MWCs THE MAJOR SOURCE OF HUMAN EXPOSURE TO PCDDs and PCDFs?

The food chain is the major source of human exposure for a large class of organics, including PCBs, pentachlorophenol, DDT, TCDD, and most pesticides (39-41). Travis and Hattemer-Frey (40) found that the food chain, especially beef and dairy products, accounts for 98% of human exposure to TCDD. Market basket studies conducted in West Germany, Canada, and Japan show that the mean background daily intake of PCDDs and PCDFs (expressed in TEDFs) in industrialized countries is about 98 pg TEDFs/day (Table 4).

Travis and Hattemer-Frey (16) evaluated risk assessments prepared for 11 proposed MWCs and estimated that the geometric mean daily intake of dioxins and furans by the *maximally exposed* individual living near a typical, modern MWC is 0.6 pg TEDFs/day. Table 4 shows that average daily intake of PCDDs and PCDFs by the maximally exposed individual living near a typical, modern MWC is due mostly to exposure to *background* PCDD/PCDF contamination. In fact, one receives 13 times more dioxin from drinking a liter (L) of milk (background concentration = 1.1 ng/L) than from breathing "dioxin-laden" air around a MWC for one day (background concentration = 4.2 ng/m^3 times 20 m^3 of air inhaled per day).

TABLE 4. TOTAL DAILY INTAKE OF PCDDs AND PCDFs (EXPRESSED IN TEDFS[a]) BY THE GENERAL POPULATION OF INDUSTRIALIZED COUNTRIES

Country	Daily intake[b] (pg/day)	Reference
West Germany	90	42
Japan	84	43
Canada	125	38
Geometric mean	98	

[a] Toxic Equivalents Dioxins and Furans

[b] Exposure estimates were adjusted to reflect the dietary intake of an individual weighing 70 kg.

These data indicate that human exposure to dioxin emitted from a typical, state-of-the-art MWC is small relative to exposure to background environmental contamination, since exposure to MWC emissions accounts for less than 1% of the total daily intake of PCDDs and PCDFs. One might question why, if measured environmental concentrations of PCDDs and PCDFs around operating MWCs are near background levels (see Tables 1 through 3), is the incremental daily intake of PCDDs and PCDFs by the maximally exposed individual living near a typical, modern MWC so small? One reason is that risk assessments prepared for proposed MWCs generally assume that only a small percentage of food is locally produced and consumed. For example, only about 5% of vegetables and produce consumed nationwide is home grown. As a result, ingestion is often not the primary pathway of human exposure. In areas of high agricultural production, however, the 0.6 pg TEDFs daily intake estimate for MWCs could be low.

CONCLUSIONS

It is currently popular among various regulatory agencies and environmental groups to blame MWCs as the major source of PCDD/PCDF input into the environment. Empirical evidence shows that MWCs are one source of PCDD/PCDF input into the environment. The same data, however, indicate that it is premature to blame MWCs as _the_ major source of PCDD/PCDF input. The relatively small contribution of MWCs to total daily intake of PCDDs and PCDFs suggests that some, as of yet unidentified, source(s) (possibly automobiles or industrial sources) are substantially contributing to background levels of dioxins and furans (16). Given the relatively high background levels of PCDDs and PCDFs, it is imperative that environmental sources of dioxins and furans be identified and quantified. Research efforts need to stress:

- Determining environmental levels of PCDDs and PCDFs in air, soil, and vegetation;
- Determining background human exposure to PCDDs and PCDFs through market basket studies;
- Quantifying the magnitude of emissions from and levels of PCDDs and PCDFs in environmental samples near facilities known to emit dioxins and furans (e.g., steel mills, copper smelting plants, MWCs etc.), so that the major contributors to the high background levels can be distinguished;
- Quantifying PCDD/PCDF emissions from vehicles that use unleaded gasoline and diesel fuel.

Given the intense public concern about dioxin, it is surprising that the EPA has yet to undertake one of the above research objectives.

The work described in this paper was funded by the U.S. Environmental Protection Agency, but this paper has not been reviewed in accordance with the Agency's administrative review policies. Therefore, the contents of this paper do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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USE OF BIOASSAY METHODS TO EVALUATE INCINERATOR EMISSIONS

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ABSTRACT

The products of incomplete combustion from incineration include complex mixtures of organics, particularly polycyclic aromatic compounds, which are present after atmospheric dilution and cooling in emissions as semi-volatile or particulate organic compounds. Combustion emissions are generally recognized as a potential cancer risk since they contain many carcinogenic and mutagenic polycyclic aromatic hydrocarbons. Analyzing such a complex mixture for the presence of even a few selected chemicals is difficult and provides risk information on only a fraction of the chemicals present. Bioassay methods, however, may be directly applied to evaluate the mutagenic and carcinogenic activity of the complex organics from combustion emissions. The Salmonella (Ames) assay was used to determine the mutagenicity associated with particles from the effluent of municipal waste combustors, from ambient air collected near a municipal waste combustor, and from the effluent of a laboratory-scale rotary kiln incinerator simulator in which polyethylene was combusted. Dose-response data were generated, and mutagenicity concentrations or mutagenic potency values were calculated.

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An environmentally acceptable means of disposal of the increasing amounts of municipal garbage has become a major problem for many city managers and engineers. The average citizen now sends 5 lbs of garbage to the dump each day, compared to 2.9 lbs/day in 1960 (1). Another source (2) estimates that this country now generates more than 450,000 tons/day of municipal solid waste. Landfills have been commonly used for disposal of municipal waste; however, in the last 20 years there have been many incidences of soil and ground water contamination. Since 1979, 3500 landfills have closed (1). The siting of new landfills is also becoming difficult in many locations due to shortage of suitable land and increasing citizen resistance to locating landfills in their areas. These problems associated with landfills have provided good incentives to utilize incineration for solid waste disposal. A further major impetus is the economic gain realized thru energy generation during municipal waste incineration (MWI).

This trend toward increased utilization of MWI, however, also mandates the development of scientific techniques needed to ensure protection of the environment and, particularly, the quality of the air that we breathe. Studies have been reported on the analysis of MWI stack emissions but usually for specific compounds or a class of compounds (3,4,5,6). Because combustion emissions or products of incomplete combustion (PICs) can literally consist of thousands of compounds, reliance on a specific analysis of only a fraction of the chemicals present would ignore the contribution of many other potentially hazardous components of the emission. This could result in an erroneous assessment of health risks associated with such emissions. Combustion emissions, therefore, are complex mixtures and this laboratory has chosen to evaluate them as such by using the short-term Ames *Salmonella typhimurium* bacterial mutagenicity assay (7) to provide a mutagenicity assessment for the whole sample. This assay was chosen because chemicals and mixtures recognized as human carcinogens are generally mutagenic in short-term tests (8) and combustion emissions from other sources have been both mutagenic in this assay and carcinogenic in rodents. It has also been reported that chemicals that are rodent carcinogens across several species and organ sites (trans-species carcinogens) are generally also mutagenic in *Salmonella typhimurium* (9). The results from studies such as these should ultimately be useful in providing information for the risk assessment of whole incinerator complex mixtures. Studies using the Salmonella assay to characterize MWI emissions have been reported (10,11,12). However, sampling and analytical techniques required for collection and sample preparation of large quantities of stack emissions are in the early stages of development.

This paper presents some of the mutagenicity results from bioassay of dichloromethane extracts of particles and/or semi-volatile organic vapors collected from (a) ambient air monitoring stations located near a municipal incinerator (13), (b) stack emissions from a different municipal incinerator, and (c) a laboratory-scale rotary kiln incinerator simulator during combustion of polyethlyene tubing. Dose-response data are used to calculate mutagenicity concentrations as revertants/m^3 for the ambient study and mutagenic potencies as revertants/µg of extractable organics in the two incinerator emission studies. None of the above field studies have been completed and only a portion of the results from each are reported here in order to demonstrate the utilization of mutagenicity bioassays.

STUDY METHODOLOGY AND RESULTS

AMBIENT STUDY

Twenty-four hour ambient air samples were collected every 12th day at four monitoring stations located around a modular mass-feed municipal waste incinerator with a burn capacity of 240 tons/day. Incinerator pollution control devices included a 4-field electrostatic precipitator (ESP) followed by a condensing heat exchanger followed by a wet scrubber (packed tower type). One or more of each following type of sampler was located at each sampling site: a) total suspended particulate (TSP) HiVol, used for collection of a mutagenicity bioassay sample on a glass-fiber filter; b) PM10 HiVol, used in part for determination of 0-10 micron particle concentrations for each sampling period; and c) PS-1 sampler, used for simultaneous collection of particles on a glass fiber filter and collection of semi-volatile organics on a polyurethane foam (PUF) plug. One site also served as a QA site and contained two TSP HiVols and two PM10 HiVols. This phase-1 data set represents the first 12 sampling periods between November 5th, 1987 and March 16th, 1988. PUF samples were collected during only one of these periods.

Filter Sample Extraction

The TSP HiVol glass fiber filters were extracted by a dichloromethane sonication procedure (14). Final extracts were concentrated by rotary vacuum evaporation and volumes adjusted to 10 mL. Triple aliquots were used for gravimetric analyses to determine extractable organic mass (EOM) concentrations. An aliquot of the remaining extract was exchanged to DMSO (14) using dry nitrogen evaporation just prior to bioassay.

Bioassay Analysis for Mutagenicity

The Ames *Salmonella typhimurium* histidine reversion assay (7) with strain TA98 was used for mutagenicity bioassay. Samples having sufficient EOM were tested at a minimum of five doses using triplicate plates with and without Aroclor-induced S9 metabolic activation (+S9 and -S9) at each dose. The +S9 data from January 16th, March 4th, and March 16th were from single plate determinations. Spontaneous reversion rates for TA98 were 25-50 colonies per plate after a 72-hour incubation. A set of positive controls was incorporated in each experiment. These controls included 2-anthramine (0.5 μg/plate) and 2-nitrofluorene (3.0 μg/plate) with and without S9, respectively. A negative control utilized in each experiment consisted of a DMSO blank.

The data were analyzed by the statistical method of Bernstein et al.(15). The slope values are reported as either revertants/μg of extractable organics or revertants/m^3 of air.

Ambient study results

TSP HiVol filters and PS-1 PUF samples from the 12 sampling periods of Nov. 5th thru March 16th were analyzed for mutagenicity. Each sampling period generated five TSP filters and five PM10 mass filters from the four ambient monitoring stations. One TSP filter was received from each of three sites, whereas the QA site generated duplicate bioassay and duplicate PM10 mass filters from the collocated samplers. Data points are missing from the graphs for sample periods for which there were insufficient amounts of extractable organics for both the +S9 and -S9 mutagenicity assays. The January 4th and February 21st TSP bioassay samples were not analyzable due to contaminated S9.

Figure 1 shows the TSP +/-S9 mutagenicity concentrations from each of the sampling periods and compares these data with the particle concentrations obtained from the collocated PM10 HiVol samplers at the four monitoring sites. The two PM10 samplers at the QA site (2nd and 3rd points for each sampling period) showed the highest concentrations of 0-10 micron particles. Likewise, the two TSP samplers from the same site gave correspondingly higher concentrations of both direct-acting (activity in the absence of S9 [-S9] microsomal activation) and indirect-acting (+S9) mutagens. This site is about 0.75 miles to the east of the incinerator and is the farthest from the stack. This site is also closest to the town center and is more heavily impacted by normal city combustion sources. The relative amount of direct-acting mutagenicity in these samples provides further evidence of significant contribution from city sources such as diesel vehicles. It is also interesting to note the general trend in particle and mutagenicity concentrations from all sites was a gradual increase from relatively low levels in November to peak concentrations in mid-January followed by a gradual decrease to low levels in mid-March.

Figure 2 demonstrates the positive correlation between the TSP indirect-acting mutagenicity and the PM10 particle concentrations. The r-squared correlation value of 0.74 would likely be further improved if the identical samplers had been available and used for this comparison.

A set of PS-1 PUF samples from one sampling period was also similarly analyzed to determine relative mutagenicity concentrations resulting from semi-volatile organics. These results (data not shown) from three sites, compared to those for the particle-bound organics, indicate that mutagenicity was associated primarily with the particles. A fourth site, however, indicated a mutagenicity concentration from semi-volatile (PUF) organics that was equal to that obtained from the filter particles. Further studies of the semi-volatile organics are in progress.

MUNICIPAL INCINERATOR STACK SAMPLING STUDY

Incinerator stack emission samples were collected by two prototype dilution samplers. The municipal incinerator used for this initial sampler testing was equipped with an electrostatic precipitator (ESP) control device only. Each sampler was designed to collect particulate samples in a manner that simulates the flue gas quenching processes occurring upon emission to the atmosphere. A 10:1 dilution with ambient air was used that promotes nucleation of vapor-phase organic compounds and condensation on existing particulate matter. A prototype source dilution sampler (SDS) was designed to collect 10 CFM stack emissions (16) and dilute in a mixing chamber with 100 CFM of heated HEPA- and charcoal-filtered outside air. Diluted particles were collected on a round (approximately 26 inch diameter) Teflon-impregnated, glass-fiber filter. A modified SASS cyclone used at the sampler inlet removed particles >2.5 microns prior to dilution and collection.

A prototype baghouse/dilution tunnel sampler, which was evaluated at the same incinerator, was designed to collect 100 CFM of stack emissions diluted with 1000 CFM of HEPA- and charcoal-filtered outside air (17). The dilute effluent was passed through a baghouse containing a fabric filter cartridge. An air jet above the cartridge was used to provide a reverse pulse of high pressure nitrogen to dislodge particles from the filter for collection in a glass bulb positioned at the bottom of the collector. This sampler was designed to collect the large quantities of sample required for chemical characterization through use of bioassay-directed fractionation and also to generate sample sizes sufficient for mouse skin carcinogenicity studies.

Sample Extraction and Bioassay

The SDS filters were cut into pieces and triple extracted by repeat sonications with dichloromethane. Extracts were concentrated, and EOMs were determined as described previously. Aliquots of remaining extract were concentrated and exchanged to DMSO prior to bioassay.

A 102-gram portion of the baghouse particle sample was Soxhlet extracted with dichloromethane. The extract was concentrated, the percent extractable mass was determined, and the sample was prepared and bioassayed as described previously.

Incinerator Stack Sample Results

Five SDS filter samples were collected during a first sampling study that contained 1.3-3.8 g of particles from 12.8-80.3 m^3 of stack emissions. Sampling period durations ranged from 56 minutes to 360 minutes. The amount of particles collected was, of course, proportional to volume of stack emissions sampled but was also dependent on burn conditions during the sampling period. The percent EOM values averaged 0.87 +/- 0.5. One sample, representing 65.3 mg of extractable organics, was bioassayed. Figure 3 shows little difference between the mutagenic potencies (revertants/microgram of EOM) of the sample in the presence of S9 (26.6 rev/µg) and the absence of S9 (19.7 rev/µg). It is interesting that the sample shows direct-acting mutagenic activity, which suggests the possible presence of direct-acting mutagens such as nitroaromatics. Further studies are in progress to elucidate the presence of such compounds.

A second sampling study, conducted approximately one month after the first study, utilized both the SDS sampler and the baghouse sampler. The purpose of this study was to compare directly the bioassay analyses for samples collected by these two prototype samplers. Water condensation was encountered during this study, however, which prohibited the baghouse sampler from dislodging particles collected on the fabric filter. An acidic wet sample was collected that contained amounts of metal ions resulting from attack on the stainless steel sampler housing by HCl in the stack emission gases. This sample, when bioassayed, proved to be toxic to Salmonella, preventing determination of sample mutagenicity. Extracts from the SDS filters collected during this same study were also found to be toxic in the bioassay system. This result was not expected because the previous mutagenicity analysis of an SDS filter sample had been successful as reported above. This second set of SDS extracts appeared to be more acidic than the first set. This condition likely caused the observed toxicity and demonstrates the need for additional research to design sampling systems for acidic stack gases.

POLYETHLYENE BURN LABORATORY STUDY

In order to study a "worst case scenario," we evaluated the organic extract of particulate emissions from a laboratory-scale rotary kiln incinerator simulator (73 kW; 250,000 Btu/h) in which polyethylene plastic was incinerated under failure mode conditions. The design of the kiln and the conditions under which the plastic was burned have been described (18). Although the kiln was operated at typical incinerator conditions (temperature, excess air, residence time) and the plastic was batch charged into the kiln to examine transient behavior typical of kilns, no afterburner or other pollution control was present on the stack. Thus, the particles obtained represent those produced under less than ideal combustion conditions. The particles were collected using the baghouse/dilution tunnel sampler described above. Particles were extracted, prepared, and bioassayed as described previously.

Figure 4 shows the mutagenic potency (in the presence of S9) of the EOM from the combustion of polyethylene in the kiln to be 18.8 rev/μg, which is similar to that of the effluent from the MWI (Figure 3). However, unlike the MWI effluent, the effluent from polyethylene combustion in the kiln was not mutagenic in the absence of S9 (Figure 4). Such a result is not unexpected because PICs formed during polyethylene combustion contain significant amounts of polycyclic aromatic hydrocarbons (PAHs) (18). These compounds generally require metabolic activation in order to be mutagenic. Thus, it is likely that some of the mutagenic activity observed from combustion of polyethylene in the kiln is due to PAHs or PAH-like compounds.

UTILITY OF BIOASSAY RESULTS

The contrasting results in the absence of S9 between the effluent from a commercially operating MWI (Figure 3) and from a laboratory-scale rotary kiln simulator burning polyethylene plastic (Figure 4) illustrate the utility of bioassays to help characterize differences between combustion emissions. Fractionation of these organic extracts by means of high performance liquid chromatography and subsequent bioassay of the fractions can be used to identify the major chemical classes that are responsible for the observed biological activity. The coupling of biological analysis with chemical analysis (bioassay-directed chemical analysis) can enhance our understanding of the chemical composition and any consequent health risks associated with incinerator emissions.

The mutagenic potency values, combined with the amount of waste burned, permits calculation of a mutagenicity emission factor. The emission factors could be compared to those from other combustion sources, such as cigarette smoke, coal-fired power plants, wood burning stoves and automobile exhaust to allow incinerator emissions to be put into perspective (19).

SUMMARY AND CONCLUSIONS

The Ames bioassay has been used successfully to determine the mutagenicity of ambient air particles collected near a municipal incinerator. This procedure will be used to provide an assessment of the contribution of the incinerator to mutagen concentrations in air through a) assessment of the relationship, if any, between tons of waste burned and mutagen concentrations at ambient air sampling sites and b) determination and comparison of mutagen concentrations in air samples collected upwind and downwind from the incinerator.

Two prototype municipal incinerator stack samplers have been built, and field testing is currently underway to evaluate their usefulness in: a) assessing the mutagenic potencies of particles and vapor-phase organics emitted by various types of municipal incinerators, b) determining the concentration of mutagens emitted by incinerators, and c) collecting sufficiently large quantities of emission particles that can be used for chemical characterization and carcinogenicity bioassays. Initial testing has encountered problems with corrosive emission gases that have attacked sampling equipment. Some stack sample extracts have also been found to be toxic to Salmonella test organisms and could not be bioassayed without fractionation to separate the toxic components from the mutagenic components. Further modification of sampling equipment and test procedures are currently being explored.

The utility of the Salmonella assay to identify differences among various incinerator emissions will be explored further by fractionating selected emissions and bioassaying the fractions. Such procedures should lead to a better understanding of the chemical composition and possible health risks associated with incinerator emissions.

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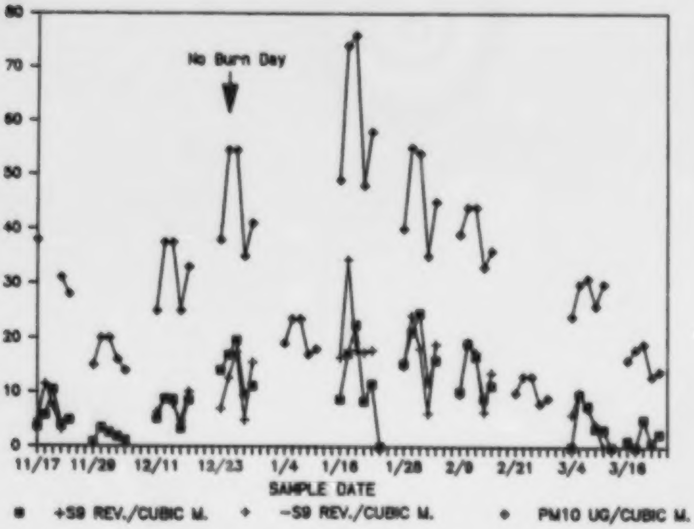

Figure 1. Mutagenicity and Particle Concentrations

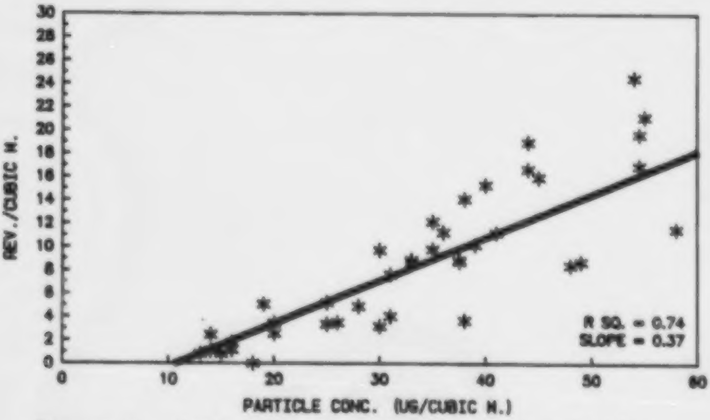

Figure 2. Mutagenicity and Particle Correlation

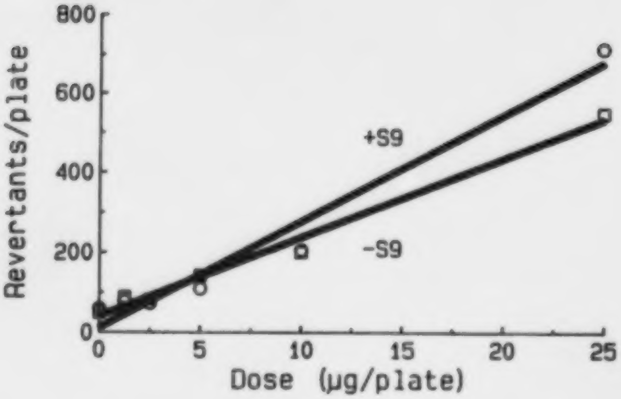

Figure 3. Mutagenicity of MWI Effluent

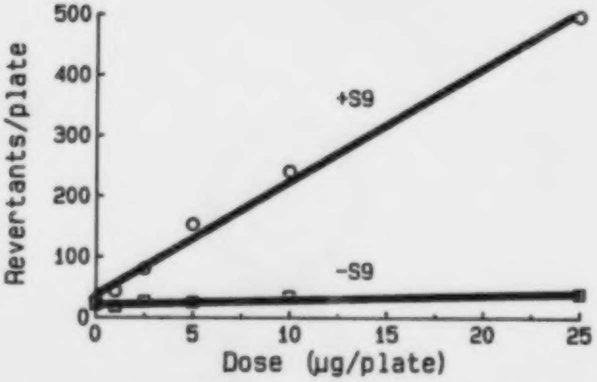

Figure 4. Mutagenicity of Rotary Kiln Effluent

SOURCE TEST RESULTS OF THE PIGEON POINT ENERGY GENERATING FACILITY, NEW CASTLE, DE

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ABSTRACT

The Pigeon Point Energy Generating Facility (EGF) fires a combination of Refused Derived Fuel (RDF) and Municipal Solid Waste (MSW). The configuration of the facility has 3 incinerators providing heat to 2 boilers on the north side, and 2 incinerators providing heat to 2 boilers on the south side. Each boiler has an ESP to control particulates.

The testing was accomplished in two phases. Phase I consisted of ESP inlet particulate, and stack particulate, SO_2, SO_3, NOx and HCl testing of all boilers. One run was accomplished during sootblowing operations. Phase II consisted of dioxin/furan, and metals (Arsenic, Beryllium, Cadmium, Chromium, Lead, Mercury, and Nickel) testing of one flue.

The test results showed the EGF to be in compliance with the permitted levels of particulate, SO_2 and NOx. The dioxin/furan and metals emissions were below those that would cause unacceptable ambient concentrations, as defined by the Pennsylvania BAT for Municipal Waste Combustion.

BACKGROUND

The original construction permit for the Energy Generating Facility (EGF) at Pigeon Point, Delaware was issued in 1984 to the owner/operator, The Crouse

Companies. The EGF commenced operations in December 1986. Shortly thereafter, the Crouse Companies filed for reorganization under Chapter 11 of the Bankruptcy Code. The EGF continued to operate under Chapter 11, and in May 1987, a through-put test was accomplished in an attempt to demonstrate that the EGF could meet its contractual obligations to the Delaware Solid Waste Authority.

In June 1987, Crouse abandoned the EGF "in place"; and the responsibility for operating the EGF fell to General Electric Capital Corporation, representing the ownership, and to Federal Insurance Company, for completion of construction. In July 1987, a temporary operating permit was issued to Vicon Recovery Systems, Inc. to operate the facility for GECC. Federal Insurance Co. hired MMR Wallace Construction Co., CSI Resources Systems, Inc., Kidde Consultants, and others to complete the construction.

After extensive rework, repair, and completion of construction, a second through-put test was accomplished in October 1987. The testing to demonstrate compliance with the conditions of the Air Pollution Control Permit was accomplished in December 1987 and January 1988. The testing contractor was Roy F. Weston, Inc. After it was determined that compliance had been demonstrated, an Operating Permit was issued in January 1988.

DESCRIPTION OF ENERGY GENERATING FACILITY

The EGF fires a combination of refused derived fuel (RDF) and Municipal Solid Waste (MSW). The RDF is supplied by the Delaware Reclamation Project which process MSW into 1) RDF (light materials: papers, plastics, etc.), 2) ferrous metals, 3) non-ferrous metals, 4)sand, 5) glass, and 6) organic residue.

There are five 120 tons per day incinerators, four boilers, and two turbine generators. The steam which is produced is used to generate electricity, and is sold to a local production facility.

The Enercon Systems, Inc. incinerators include a six hearth primary combustion chamber and a secondary combustion chamber. (See Figure 1). The configuration of the facility has three incinerators feeding a tertiary duct which provides heat to two boilers on the north side, and two incinerators feeding a tertiary duct which provides heat to two boilers on the south side. Each boiler has an ESP to control particulates. Cleaned flue gas is recirculated and used for incinerator over and under fire, as well as to control the temperature of the gas entering the boilers. (See Figure 2).

These incinerators are of similar design as the Vicon Recovery Systems Pittsfield, MA facility, which was tested extensively for dioxins and furans under a program sponsored by the New York State Energy Research and Development Authority. However, the EGF units fired a 5 RDF to 1 MSW (by weight) mixture during the testing, while Pittsfield fires 100% MSW.

PARTICULATE

The particulate limit for the EGF is .05 grains per dry standard cubic foot corrected to 12% CO_2. The particulate testing was accomplished at the inlet to the ESP's and in the stacks for all 4 boilers. EPA reference Method 5 was used. At the ESP inlets the sampling train was modified to include an in-stack tared glass fiber thimble and thimble holder, placed between the nozzle and the probe. This was done to reduce the loading to the Method 5 filter, which was also used.

As can be seen in Figure 3, the duct configuration at the ESP inlets is less than ideal, both for stack testing and introducing gases into an ESP. The direction of flow was determined using a 3 hole pitot to measure the angles of pitch and yaw. There are straightening vanes in duct, and the flow was acceptable. The variation of the flow within the duct was analyzed by a 3 way analysis of variance. The flow on the north side, units 1 and 2, was considered to be distributed evenly; while the flow on the south side, units 3 and 4, was not. It appears that the straightening vanes are more effective on the north side, which has the greater flow. On the soutn side, one of the six sampling ports was inaccessible for each unit because of a straightening vane.

Prior to the compliance tests, the particulate emissions were measured during each of the throughput tests (May and October 1987). The results are summarized below and with one exception, have been well in compliance with the particulate limit of 0.05 grains/dscf corrected to 12% CO_2.

Unit #	May 1987 Throughput	October 1987 Throughput	December 1987 Compliance
1	.004	.0012	.0029
2	.004	.0011	.0015
3	.009	.0014	.0063 (.0018)*
4	.008	.2136	.0053

* Average of 2 runs, 1 run suspected of scraping side of flue.

During the compliance tests, the inlets and stacks were tested simultaneously for the north side, units 1 and 2, and the south side, units 3 and 4. Sootblowing was accomplished during one of runs for each of the units. The results are summarized as follows.

Unit		No Sootblowing*			Sootblowing**		
		In	Out	Eff	In	Out	Eff
#1	GR/DSCF @ 12% CO_2	.7656	.0017	.998	1.4889	.0052	.997
	LBS/HR	193.9	0.3		338.4	1.0	
#2	GR/DSCF @ 12% CO_2	.5292	.0017	.997	1.9949	.0012	.999
	LBS/HR	150.0	0.3		550.4	0.2	
#3	GR/DSCF @ 12% CO_2	.4668	.0023**	.995	1.7529	.0014	.999
	LBS/HR	87.2	0.3		394	0.2	
#4	GR/DSCF @ 12% CO_2	.4291	.0069	.984	.4572	.0021	.995
	LBS/HR	94.0	0.6		114.3	0.2	

* Average of 2 runs - ** 1 run

Because a portion of the cleaned flue gas is recycled, the ESP efficiencies are based on the grain loading, not on mass rates.

The effect of sootblowing is obviously to increase the loading to the ESP's; although it is not clear why unit #4 did not increase as much as the others. A three way analysis of variance was used to reach the following conclusions regarding the inlets to the ESPs.

1) The concentration of particulate was greater for the north side, units 1 and 2. While the mass rates were expected to be greater, because of the additional incinerator, the concentrations were expected to be the same. However, at the ESP inlets, the dry standard cubic feet per pound of RDF/MSW fired were not consistent. On the North Side, there were 176 dscf/#RDF/MSW, and on the South Side 191. This would support the higher particulate concentration on the north side.

2) The ESP inlet loadings are not the same for the units.

3) ESP loading is greater when soot is being blown.

The particulate emission data measured at the stacks were also analyzed using a three way analysis of variance with the following conclusions.

1) Particulate emissions are the same for each side of the plant.

2) Particulate emissions are the same for each unit.

3) There is no difference in particulate emissions when there is sootblowing.

During the test period, operating data was recorded for all the ESP's. The secondary power and spark rates were used for a 3 way analysis of variance to conclude that the operation of the ESPs were independent of each other. Table 1 is a summary of baseline data for the ESP's. The ESP's were operating in a manner such that an emission limit of 0.015 grains/DSCF @ 12% CO_2 can be expected. The k coefficient, which is necessary for estimating ESP performance, has an average value of .22, and a standard deviation of .097. The 95% confidence limits are .16 to .28.

GASES

The operating permit for the ERF establishes limits for SO_2 (4.6 pounds per ton fired), NO_x (3.0 pounds per ton fired), and CO (50 ppm). These gases, and HCl, CO_2, O_2 and H_2O were measured as a part of the Phase I compliance testing, as well as during the throughput testing. The SO_2 and SO_3 were determined using EPA Reference Method 8. The HCl was measured using an EPA Method 6 sampling train with .2N NaOH in the 1st 3 impingers. The HCl analyses were done according to the colormetric, automated, ferricyanide AAII procedure specified in EPA Method 325.2. Oxides of Nitrogen were determined using EPA Reference Method 7A. EPA Reference Method 3 was used to determine O_2 and CO_2.

The Method 3 bag sample were analyzed for CO using a gas filter correlation analyzer (TECO Model 40) calibrated against known CO gas standards as per EPA Reference Method 10. The results are summarized below.

Item	Unit #	Throughput 5/87	Throughput 10/87	Compliance 12/87
SO_2 (lbs/ton)	1	3.91	3.13	4.77
	2	3.61	2.27	4.77
	3	3.98	4.06	4.0
	4	5.82	4.06	4.0
NO_x (lbs/ton)	1	6.85	<1.91	2.34
	2	6.82	<1.91	2.34
	3	4.85	<1.95	<2.04
	4	7.50	<1.95	2.04
CO (ppm)	1			4
	2			4
	3			5
	4			6
HCl (ppm)	1		590	412
	2		327	355
	3		414	337
	4		343	388
CO_2 (%)	1	10.0	8.6	8.5
	2	8.7	8.6	7.8
	3	7.7	8.9	8.2
	4	6.3	8.9	8.2
O_2 (%)	1	9.6	9.5	11.0
	2	11.8	9.1	11.7
	3	11.0	11.0	11.3
	4	13.2	10.8	11.2
H_2O (%)	1		14.8	12.0
	2		14.8	12.4
	3		13.2	12.2
	4		12.8	12.5
SO_3 (ppm)	1			15
	2			13
	3			16
	4			15

The SO_2 values have varied over the three tests. While the North Side of the plant was above the permitted level of 4.6 pounds per ton, the average of the North and South sides is in compliance.

There were marked improvements in the NO_x emissions as a result of the rework accomplished during the summer of 1987. The units are well in compliance with the 3.0 lbs/ton limitation. During the test the average temperature of gas exiting the secondary chamber ranged from 1796°F to 1872°F.

The CO emissions were well below the permit limitation.

The application to construct was based on 179 pounds per hour of HCl being emitted from the plant. The results of the compliance test are 195 pounds per hour.

The application to construct indicated that the ESP was intended to be operated at 6%-7% oxygen. The actual levels, as measured at the ESP inlet and the stack were considerably higher. The oxygen levels are tabulated below.

Unit	Run	O_2 @ Incinerator Outlet	O_2 @ ESP Inlet	O_2 @ Stack
1	1	7.1	10.2	10.9
	2	7.0	10.0	11.6
	3	7.0	10.1	11.0
2	1	7.1	10.2	11.6
	2	7.0	10.3	11.7
	3	7.0	10.2	11.7
3	1	7.4	10.3	11.3
	2	7.7	9.9	11.2
	3	7.9	10.4	11.5
4	1	7.4	9.7	11.9
	2	7.7	9.4	10.9
	3	7.9	9.7	11.2

50% excess air is indicated at the outlet of the incinerator, 90% at the inlet of the ESP's and 120% at the stack. Outside air is used to prevent corrosion of isolation dampers located at the exit of each incinerator secondary chamber, entrance to each boiler, and each Flue Gas Recirculation fan. On the north side there are 7 such dampers and 6 on the south side, in addition each ESP Field uses outside air to protect the insulators.

The results of the analysis of the EPA Audit samples of SO_2 and NO_x are summarized below. The results were with the acceptable range.

	EPA	Test Contractor	% difference
SO_2	760	736.2	-3.1%
	1630	1567.6	-3.8%
NO_x	1000	1006.7	+0.7%
	445	452.2	+1.6%

DIOXINS/FURANS

Currently, limits for dioxins/furans in the stack gases have not been established. The test protocol for dioxins and furans required that the sampling be accomplished following the procedures described in the "Environmental Standards Workshop Protocol (December 1984 Draft)", and that the analyses be accomplished in accordance with the "Analytical Procedures to Assay Stack Effluent Samples and Residual Combustion Products For Polychlorinated Dibenzo-P-Dioxins (PCDD) and Polychlorinated Dibenzo-Furans (PCDF)" prepared by Group C-Environmental Standards Workshop, September 18, 1984, revised December 13, 1984. High resolution gas chromatography was specified. The front and back halves of the train were to be quantitated seperately.

Due to the lack of availability of quality assurance audit samples, the protocol required that the recovered samples be split and analyzed by two laboratories. Enseco's California Analytical Laboratory accomplished the full analyses and the New York State Department of Public Health Laboratory accomplished the quality assurance analyses. Table 2 shows the results of the analyses from the two laboratories. We considered the replication to be acceptable based on a Wilcoxin Paired Sample statistical test of the back-half results of runs 1 and 3. The relative differences between the two laboratories for these runs were 31.3% and 22.6%.

Table 3 shows the emissions in ng/dscm, and as 2378 TCDD toxic equivalents. Using the modeling data associated with the application to construct, a maximum annual concentration 0.004 x 10^{-6} ug/m^3 is predicted. This is almost 2 orders of magnitude less than the action limit of 0.3 x 10^{-6} ug/m^3, specified in the State of Pennsylvania's BAT for Municipal Waste Combustion. The concentrations in the Fly Ash are also shown in Table 3.

Consistent with the low particulate emissions, 98.6% of the Dioxins and Furans were collected in the back half of the train as gases.

There has been some conjecture that dioxin/furan emissions from incinerators firing RDF may be greater than those firing MSW only. The Pigeon Point incinerators are of similar design as the Vicon units in Pittsfield, MA which were studied extensively under a research project sponsored by the New York State Energy Research and Development Authority. Table 4 shows the EGF dioxin/furan emissions with those of Pittsfield and other Facilities. The EGF emissions are comparable with the Pittsfield Phase II results. It does not appear that the dioxin/furan emissions are sensitive to the type of fuel, MSW or RDF, being fired.

We were also interested in whether the dioxin/furan content of the fly ash could provide insight into the stack gas, and how the EGF stack gas compared to other facilities. The distributions of the homologues/cogeners were the criteria used. The distributions in the stack gases were based on the ng/m^3 data.

Table 5 shows the distribution of dioxins/furans in the Stack Gas and Fly Ash of the EGF, and in the stack gases from other facilities. Based on a

Spearman Rank Correlation, the distributions in the ESP Stack Gas and Fly Ash can be considered to be correlated. It would appear that insights into the distribution of dioxins/furans in the stack gas can be obtained from analyses of the fly ash.

Table 6 is a summary of Spearman Rank Correlations of the distributions from all the facilities. There are 15 correlations out of 25 opportunities. It does not appear that the distribution of dioxins/furans will necessarily be consistent from one facility to the next. The differences in the distribution may be due to differences in the composition of the MSW being fired, the type of incinerator, or the type of control device.

METALS

Currently, permit limits have not been established for metals in the flue gases. Metals (Arsenic, Beryllium, Cadmium, Chromium, Chromium +6, Lead, Nickel, and Mercury) were measured in both the stack gases and the ESP fly ash. The Metals (except for Chromium +6) in the stack gases were sampled by using a Modified Method 12 train. The Method 12 train was modified to have the third impinger contain freshly acidified 4% potassium permanganate solution. The Chromium +6 was sampled in accordance with an EPA draft method, which is essentially a Method 5 train. The analyses were conducted as follows: 1) Arsenic and Lead by graphite furnace atomic absorption spectrophotometry, 2) Beryllium, Cadmium, Chromium, and Nickel by ICP atomic emission spectrophotometry, 3) Chromium +6 by the diphenylcarbazide (DPC) spectrophotometric method described in the EPA draft CR +6 sampling and analysis method, and 4) Mercury by a manual cold vapor technique which is designed to oxidize inorganic as well as organic Mercury vapor during the digestion process; the mercury was purged from the sample by reduction with stannous sulfate, and measured by a spectrophometer. The ESP ash removal system was operated in such a manner that the fields of the ESP could be sampled individually. These samples were analyzed in accordance with EPA Method 6010 (ICP)

Because we did not have any experience with the multi-metals approach to stack sampling and analysis, it was required that an audit sample be analyzed prior to the test run, and a second be analyzed with the test run. The audit samples were metals on a filter and vials containing mercury solution. Both were supplied by the EPA. A portion of the Mercury solution was added to the filter digest, and a portion was analyzed by itself. The results are summarized below.

Audit Media	Metal	Pre-Test (% Difference)	Test (% Difference)
Filter	Chromium	+2.4	+20.8
	Arsenic	-2.6	+4.0
	Cadmium	*	*
	Lead	+1.0	+6.4
	Mercury	-13.5	+9.2
	Beryllium	-11.6	+7.6
	Nickel	-0.8	+17.4

Vial	Mercury	-11.5	+12.5

Cadmium was reported at low levels, but was not present on the filter.

We considered the audit results to be acceptable.

The metals results are shown in Table 7. The blank metals train had Cadmium results higher than the stack train, and therefore, Cadmium was not reported. The contractor estimated the Cadmium results by analyzing the filter and acetone rinse residue from the particulate runs for both Cadmium and Lead. On the basis that the Lead results from that analysis were consistent with the Lead results from the metals testing, Cadmium was estimated to be comparable also. That is the basis for the 9 ug/m^3 estimate shown for Cadmium.

Based on the stack concentrations, and the modeling data used for the application to construct, the projected ambient concentrations do not appear to be at a level which should cause concern. Listed below are the projected maximum annualambient concentrations and the ambient concentrations listed in the Pennsylvania BAT for Municipal Waste Combustion.

Metal	Maximum Annual Projected Ambient (ug/m^3)	PA BAT (ug/m^3)
Arsenic	.4 x 10^{-5}	.23 x 10^{-3}
Beryllium	.2 x 10^{-5}	.42 x 10^{-3}
Cadmium *	.6 x 10^{-4} *	.56 x 10^{-3}
Chromium	.12 x 10^{-3}	
Lead	.7 x 10^{-3}	.50
Nickel	.2 x 10^{-3}	.33 x 10^{-2}
Mercury	.0018	.08
Chromium +6	.11 x 10^{-4}	.83 x 10^{-4}

* Estimate

The fly ash data was examined by several statistical methods, both parametric and non-parametric. The non-parametric methods were the Mann-Whitney test for sootblowing vs non-sootblowing, and the Kruskal-Wallis single factor analysis of variance (ANOVA) by ranks for differences between ESP fields and test day. Non-parametric statistical methods do not make any assumptions regarding the distribution of the sample's population, and therefore may be more appropriate than parametric methods which assume that the sampled population is normally distributed. However, if parametric methods are appropriate, they will be more powerful than non-parametric (i.e. non-parametric methods have a greater probability of not rejecting a null hypothesis when it is false)[1]. The parametric methods used were single factor analysis of variance (ANOVA), two way ANOVA, and three way ANOVA. Table 8 is a summary of the results. My conclusions regarding this data are:

1) The statistical results are not entirely consistent.

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[1] Biostatistical Analysis, By Jerrold H. Zar.

2) There is no difference in the metals concentration of the fly ash whether sootblowing is being accomplished or not.

3) The metals concentration vary by ESP field for Cadmium and Lead, and may vary for Arsenic. The concentrations are greater in the third field.

4) The Lead concentrations varied by day. Beryllium, Cadmium and Nickel concentrations were more variable than other metals.

An objective of accomplishing metals analyses of the fly ash in conjunction with the stack tests was to determine if they are correlated. Based on a Spearman Rank Correlation, they are not. This is not surprising considering the variability of the vapor pressures of the metals and their products of combustions (Oxides and Chlorides). There is an indication that at the stack gas temperature, the gas phase is predominant. Table 9 shows the fly ash concentrations, predicted stack gas particulate concentrations (Based on fly ash with .0015 grains/DSCF), and measured concentrations. The measured concentrations are greater than those predicted based on particulate alone. Thus, at the stack gas temperature of 375°F, most of the metals appear to be in the gas phase. However, as discussed previously, an analysis of a particulate filter indicates that Lead was a particulate at the filter temperature of 225°F-275°F. This appears to support the idea that metals' emissions will be reduced by quenching of the flue gases prior to the particulate control device.

PLANT OPERATIONS

A number of incinerator and boiler operating parameters were recorded during the compliance test. These are recapped in Tables 10 & 11. The oxygen and temperature at the exit of the secondary incinerator were within acceptable ranges. During the dioxin/furan and metals testing, incinerator #3 had bridging problems which caused that incinerator to have positive pressure intermittently. This was not a desireable operating condition.

Table 12 compares compliance test data with throughput test data and design data. During the particulate and gas compliance tests there were considerably more stack gases per pound of MSW/RDF fired than during the throughput test. There was also more steam generated per unit MSW/RDF fired.

CONCLUSION

The compliance testing of the Pigeon Point Energy Generating Facility demonstrated that the facility was in compliance with the permitted levels of particulate, SO_2 and NOx.

The dioxin/furan and metal emissions were below those that would cause unacceptable ambient concentrations, as defined by the Pennsylvania BAT for Municipal Waste Combustion.

The operation of the boiler and incinerators were considered to be normal. The particulate removal efficiencies of the ESP's exceeded 99%.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

ACKNOWLEDGEMENTS

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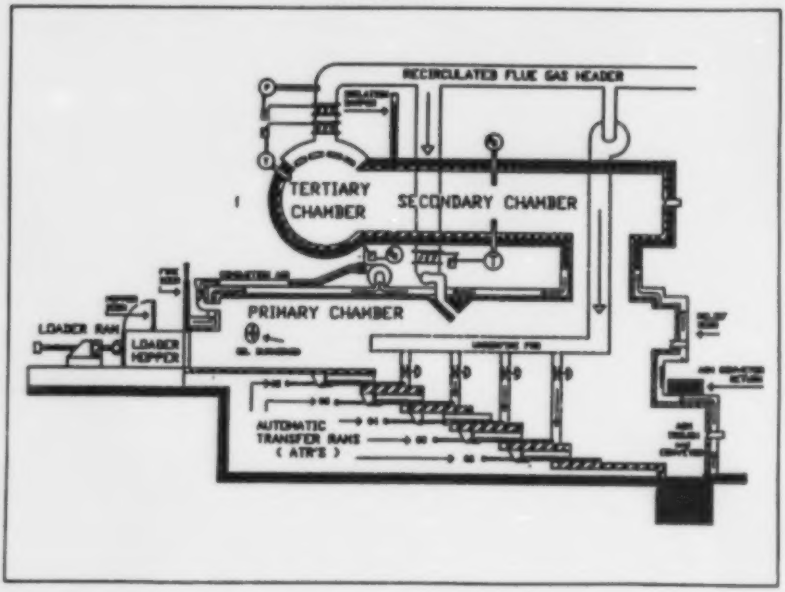
General Electric Capital Corporation
MMR/Wallace Construction Company, Inc.
CSI Systems, Inc.
Roy F. Weston, Inc.
Enseco California Analytical Laboratory.
New York State Department of Public Health Laboratory
Technical Services Section, Delaware Division of Water Resources

Their efforts in connection with the test program are greatly appreciated.

Figure 1

Incinerator Schematic


RECIRCULATED FLUE GAS HEADER
TERTIARY CHAMBER
SECONDARY CHAMBER
PRIMARY CHAMBER
LOADER RAM
LOADER HOPPER
AUTOMATIC TRANSFER RAMS (ATR'S)

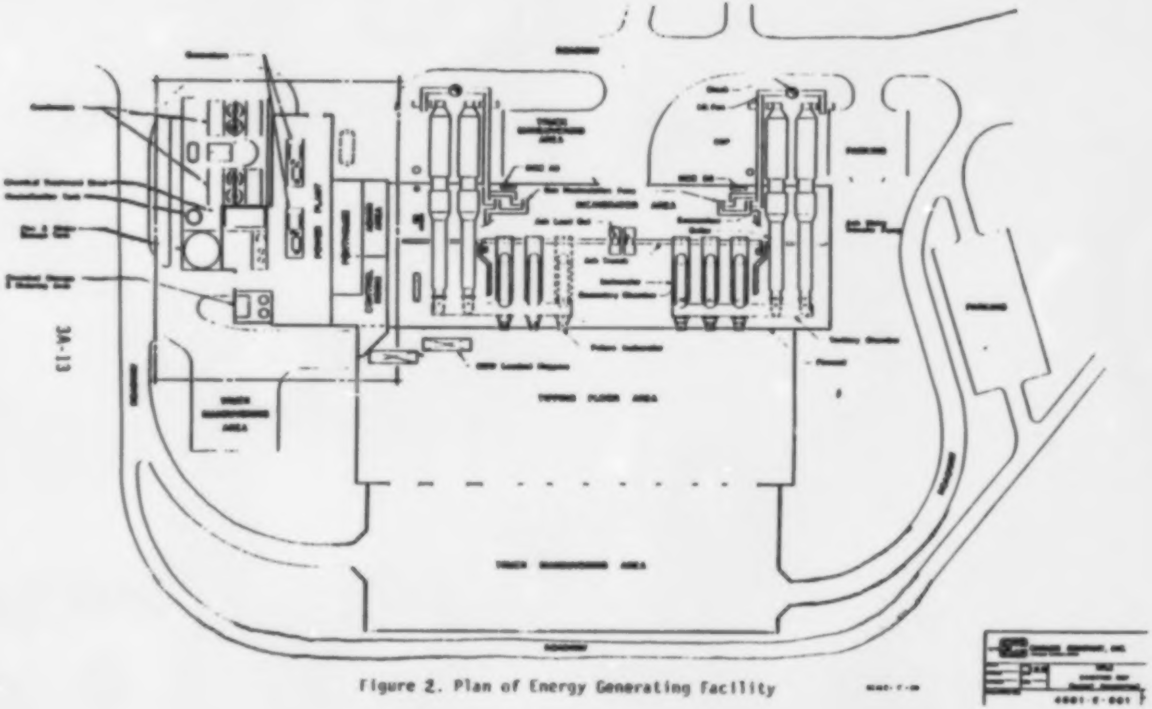


Figure 2. Plan of Energy Generating Facility

Figure 3

ESP Inlet Ducting

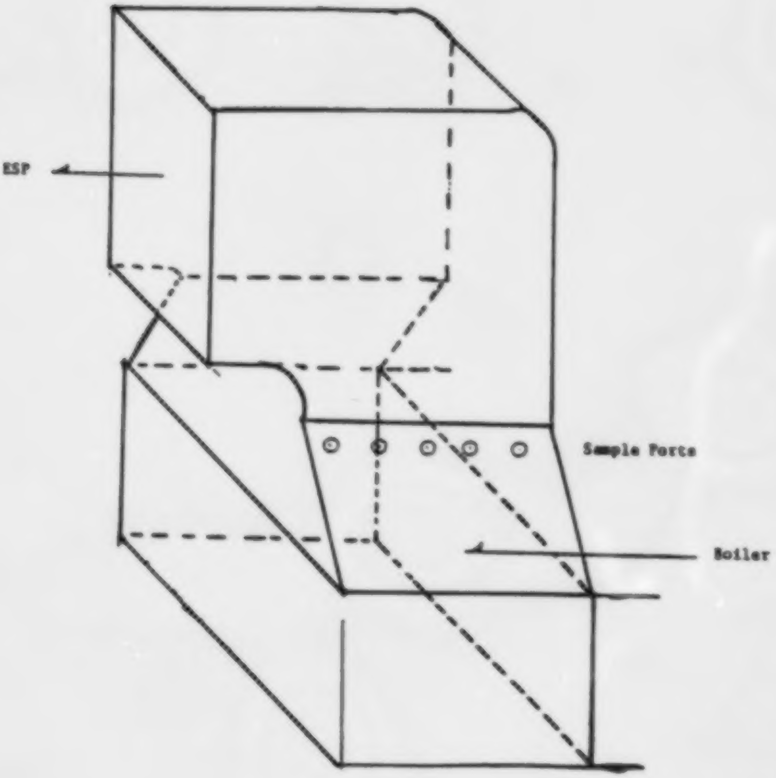

ESP
Sample Ports
Boiler

TABLE 1

Baseline Data For Inspections

Item	ESP #1			ESP #2			ESP #3			ESP#4		
	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3
ESP Gas Temp. (^{o}F)	434	430	435	436	422	435	404	393	381	418	421	402
Total Power (Watts)	21900	28600	16900	37500	32200	31700	29200	28900	28600	47900	47800	47900
Gas Flow (acfm)	67300	74600	70300	84100	80100	88900	57000	61600	62200	64900	68500	61600
Specific Power (Watts/ Kacfm)	325	383	240	446	402	357	512	469	457	738	698	778
Particulates (gr/dscf @12 % CO_2) IN	.6978	.8933	1.4887	.4883	1.9949	.5701	.4168	1.7529	.5168	.4172	.4572	.4409
OUT	.0017	.0019	.0052	.0013	.0012	.0021	.0154	.0016	.0023	.0080	.0021	.0058
Penetration (1-EFF)	.0024	.0021	.0035	.0027	.0006	.0037	.0369	.0008	.0045	.0192	.0065	.0132
*K Factor 15	.31	.27	.39	.22	.31	.26	.11	.25	.20	.09	.13	.09
			SOOT BLOW		SOOT BLOW			SOOT BLOW			SOOT BLOW	
CO_2 ESP Inlet	9.2	9.9	9.1	9.1	9.5	9.1	9.0	9.6	9.0	9.4	10.0	9.4
Stack	8.4	8.7	8.5	7.6	7.9	8.0	8.1	8.2	8.2	7.9	8.5	8.2
O_2 ESP Inlet	10.2	10.2	10.1	10.2	10.3	10.2	10.3	9.9	10.4	9.7	9.4	9.7
Stack	10.9	11.6	11.0	11.6	11.7	11.7	11.3	11.2	11.5	11.9	10.6	11.2
Specific Power Required for Compliance @.05 gr/dscf	142	178	145	173	198	156	321	237	195	393	284	403
Specific Power Required for .015 gr/dscf	206	252	196	264	263	233	512	317	295	616	438	626

*Calculated from: $P=e^{-.06K\,(Pc/v)}$

or $K=\frac{\ln P}{-.06\left(\frac{Pc}{v}\right)}$

After EPA -340/1-79-007 Inspection Manual for Evaluation of ESP Performance.

TABLE 2
SDF TESTING

DIOXIN/FURAN SUMMARY OF SPLIT SAMPLES
NANO GRAMS

HOMOLOGUE/ CONGENER	METHOD BLANK		METHOD BLANK		BLANK COMP.		LAB BLANK		RUN #1 FRONT		RUN #1 BACK		RUN #3 FRONT		RUN #3 BACK	
	MT	CA	MT	CA	MT	CA	MT	CA	MT	CA	MT	CA	MT	CA	MT	CA
2378 TCDD	LT	ND	LT	ND	XX	ND	LT		LT*	.073	0.3	.310	LT	ND	LT	.310
12378 PCDD	LT	ND	LT	ND	LT	ND	LT		LT	ND	1.2	1.6	LT	ND	1.8	.850
123678 HxCDD	LT	ND	LT	ND	LT	ND	LT		LT	ND	1.2	** (.820)	LT	ND	1.6	1.300
123789 HxCDD	LT	ND	LT	ND	LT	ND	LT		LT	ND	1.2	1.1	LT	ND	1.7	1.300
123478 HxCDD	LT	ND	LT	ND	LT	ND	LT		LT	ND	0.8	ND	** (.670)	ND	1.2	.860
1234678 HpCDD	LT	ND	LT	ND	LT	ND	LT		LT*	.550	11.0	11.0	LT	1.4	17.0	15.728
12346789 OCDD	LT	ND	LT	.125	LT	.16	LT		5.7	4.6	27.0	43.0	2.5	1.470	35.0	36.570
2378 TCDF	LT	ND	LT	ND	XX	ND	LT		LT*	.027	9.8	2.4	LT*	.017	10.0	2.1
12378 PCDF	LT	ND	LT	ND	LT	ND	LT		LT	ND	3.9	3.0	LT	ND	4.4	3.6
23478 PCDF	LT	ND	LT	ND	LT	ND	LT		LT	ND	7.0	2.3	LT	ND	9.3	3.4
123478 HxCDF	LT	ND	LT	ND	LT	ND	LT		LT*	.069	4.9	2.7	LT	ND	9.3	8.1
123678 HxCDF	LT	ND	LT	ND	LT	ND	LT		LT	ND	2.0	3.1	LT	ND	4.9	3.9
234678 HxCDF	LT	ND	LT	ND	LT	ND	LT		LT	ND	4.3	3.8	LT	ND	7.3	1.2
123789 HxCDF	LT	ND	LT	ND	LT	ND	LT		LT	ND	LT	1.5	LT	ND	LT	5.9
1234678 HpCDF	LT	ND	LT	ND	LT	ND	LT		LT	ND	18.0	20.0	LT*	.17	35.0	33.002
12346789 OCDF	LT	ND	LT	ND	LT	ND	LT		LT*	1.3	6.0	12.0	0.5	.278	13.0	16.980
TOTAL TCDD	LT	ND	LT	ND	XX	ND	LT		LT	ND	4.0	4.4	LT	ND	3.7	6.413
TOTAL PCDD	LT	ND	LT	ND	LT	ND	LT		LT	ND	7.0	5.0	LT	ND	12.0	9.590
TOTAL HxCDD	LT	ND	LT	ND	LT	ND	LT		LT	ND	9.9	11.0	LT	ND	13.0	20.89
TOTAL HpCDD	LT	ND	LT	ND	LT	ND	LT		LT*	1.9	19.0	21.0	LT	2.2	29.0	28.3
TOTAL TCDF	LT	ND	LT	ND	XX	ND	LT		LT*	0.65	71.0	65.0	LT*	.031	62.0	61.4
TOTAL PCDF	LT	ND	LT	ND	LT	ND	LT		LT	ND	63.0	79.0	LT	ND	86.0	97.363
TOTAL HxCDF	LT	ND	LT	ND	LT	ND	LT		LT	ND	25.0	40.0	LT	ND	44.0	68.930
TOTAL HpCDF	LT	ND	LT	ND	LT	ND	LT		LT	.069	25.0	29.0	LT*	.171	51.0	60.9

ND = Not Detected (CA)
LT = Less Than Detection Limit (MT)
XX = Interference Prevented Use of Internal Standard
*CA= Result Less Than MT Detection Limit
** = Results Reported as ND, ( ) is Maximum Possible Concentration

TABLE 3
PIGEON POINT RRF
TOTAL DIOXIN & FURAN RESULTS

LABORATORY REPORT (total ng)		TEF	STACK ng/dscm	EQUIV	FLY ASH ng/g	EQUIV
TCDF	OTHER	0.001	12.125	0.0121	4.1	0.0041
	2378	0.1	0.577	0.0577	0.25	0.0250
PeCDF	OTHER	0.001	15.628	0.0156	4.8	0.0048
	12378	0.1	0.707	0.0707	0.52	0.0520
	23478	0.1	0.649	0.0649	0.43	0.0430
HxCDF	OTHER	0.001	8.385	0.0083	3.6	0.0036
	123478	0.01	0.705	0.0071	0.48	0.0048
	123678	0.01	0.757	0.0076	0.46	0.0046
	234678	0.01	0.398	0.004	0.68	0.0068
	123789	0.01	0.833	0.0083	0.13	0.0013
HpCDF	OTHER	0.00001	3.47	0.0000	2.5	0.0000
	1234678	0.0001	5.52	0.0006	4.3	0.0004
	1234789	0.0001	0.576	0.0001	0.83	0.0001
OCDF		0	2.993	0.0000	3.1	0.0000
TOTAL FURANS			53.319	0.2605	26.18	0.1505
TCDD	OTHER	0.01	0.816	0.0082	0.4	0.0040
	2378	1	0.066	0.0660	0.054	0.0540
PeCDD	OTHER	0.005	1.111	0.0056	1.6	0.0080
	12378	0.5	0.266	0.1330	0.22	0.1100
HxCDD	OTHER	0.0004	2.491	0.0010	3.44	0.0014
	123478	0.04	0.18	0.0072	0.14	0.0056
	123678	0.04	0.264	0.0106	0.34	0.0136
	123789	0.04	0.27	0.0108	0.25	0.0100
HpCDD	OTHER	0.00001	2.552	0.0000	5.1	0.0001
	1234678	0.001	3.133	0.0031	6.4	0.0064
OCDD		0	8.019	0.0000	11	0.0000
TOTAL DIOXINS			19.168	0.2454	28.944	0.2130
TOTAL 2378 TCDD EQUIVALENTS			ng/dscm	0.506	ng/g	0.364
			ng/s	7		

TABLE 4

Comparison Of RRF With Other Plants

Plant	Sampling Location	PCDD (ng/dscm) Total	PCDD (ng/dscm) TCDD	PCDF (ng/dscm) Total	PCDF (ng/dscm) TCDF
RRF	Stack	20	.9	53	12.7
Pittsfield (1)	Phase I				
	4A-Boiler Inlet	13	6	12	2
	4B-Boiler Inlet	18	9	590	120
	Phase II				
	Tertiary Duct	5.8	.60	31	7.1
	Boiler Outlet	22	2.0	54	16
	Stack	69	3.3	100	19
Plant 1 (1)	After ESP	3200	490	5900	2500
Plant 2 (1)	" "	35	2	82	16
Plant 3 (1)	" "	2300	230	11,000	1100
Plant 4 (1)	" "	6000	690	8300	1900
Plant 5 (1)	After Quench & ESP	2400	170	2500	410
Plant 6 (1)	" "	790	100	820	290
Plant 7 (1)	" "	2700	750	6100	2600
Prince Edward Island (1)	After boiler	100	3	120	18
Sheridan Ave. New York (2)	Stack	15.7		37.1	
Oneida, NY (2)	Stack	99.1		88.4	
Marion Co. OR(3)	Stack-Compliance	1.1		.8	
	Start-up	2.2		9.6	
	Shut-down	1.2		2.3	
Tulsa, OK (3)	Stack	18.0	.101	15.5	2.905
North Andover(3)	Stack	81	1	262	16

1) From Table 7-4: Results of the Combustion and Emissions Research Project at the Vicon Incinerator Facility in Pittsfield, MA (New York State Research and Development Authority).

2) From New York Department of Environmental Conservation Reports.

3) From U.S. EPA Reports

TABLE 5

RDF Dioxin/Furan

% DISTRIBUTION

	RDF STACK	RDF FLY ASH	MARION COUNTY OR	TULSA OK	OCCI-DENTAL	WEST-CHESTER	NORTH ANDOVER
Other TCDF	16.5	6.9	9.8	12.8	12.3	27.7	16.9
2378 TCDF	0.8	0.4	10.7	4.4	0.3	2.2	5.7
Other PeCDF	21.3	8.1	1.2	5.7	19.1	11.2	9.7
2378 PeCDF	1.9	1.6	1.6	4.0	3.1	6.3	4.3
Other HxCDF	11.4	6.0	0	1.1	7.3	18.0	1.7
2378 HxCDF	3.7	3.9	1.1	0.2	3.7	2.0	2.6
Other HpCDF	6.7	4.2	0	2.3	0.6	10.4	10.0
2378 HpCDF	0.3	8.6	1.1	0.7	3.9	—	—
OCDF	8.1	5.2	2.1	2.2	0.9	.4	12.2
Total Furans	72.8	43.9	27.9	45.3	51.2	76.1	67.3
Other TCDD	1.1	0.7	7.2	4.4	5.2	2.6	3.7
2378 TCDD	0.1	0.1	5.2	0.3	0.2	0.3	0.3
Other PeCDD	1.5	8.9	2.8	8.1	9.4	2.1	5.5
2378 PeCDD	0.4	0.4	0.6	0.5	1.0	0.7	0.3
Other HxCDD	3.4	5.7	5.5	13.3	12.7	3.3	4.0
2378 HxCDD	1.0	1.3	1.5	1.5	1.6	0.6	0.9
Other HpCDD	3.5	10.1	2.9	6.5	14.9	5.5	10.3
2378 HpCDD	0.3	10.7	8.8	6.4	—	—	—
OCDD	12.0	18.4	37.5	10.1	7.5	5.3	7.7
Total Dioxin	27.3	56.2	72.1	55.0	48.3	23.9	32.7
Toxic Equivalent EPA	.548		.108	.701			0.2
(ng/dscm) ETE	872				66.1	4.16	

* Stack Gases Only

TABLE 6

DISTRIBUTION OF DIOXIN/FURAN CORRELATION MATRIX

FACILITY FUEL CONTROL	RDF RDF/MSW ESP	MARION CO. MSW DRY SCRUBBER W/BAGHOUSE	TULSA MSW ESP	OCCI-DENTAL RDF ESP	WEST CHESTER MSW ESP	NORTH ANDOVER MSW ESP
RDF	:::::::::	NO	NO	YES	YES	YES
MARION CO.	NO	:::::::::::::	YES	NO	NO	NO
TULSA	NO	YES	::::::::	YES	NO	YES
OCCIDENTAL	YES	NO	YES	:::::::::	YES	NO
WEST CHESTER	YES	NO	NO	YES	:::::::::	YES
NO. ANDOVER	YES	YES	YES	NO	YES	:::::::::

Summary of Correlations

RDF	3
Marion Co.	1
Tulsa	3
Occidental	3
West Chester	2
No. Andover	3
TOTAL	15

TABLE 7

ESP METALS SUMMARY

METAL	STK CONC. ug/m3	FLY ASH (ppm) ESP FIELD	NON 1/5/88	SOOT 1/6/88	BLOW 1/7/88	 1/5/88	SOOT 1/6/88	BLOW 1/7/88
ARSENIC	0.6	A	38.46	77.7	41.38	37.62	42.7	51.69
		B	44.51	54.39	48.65	48.73	73.5	
		C	64.61	97.64	71.61	70.30	86.0	63.68
BERYLLIUM	0.3	A	0.90	0.84	0.86	3.38	0.87	0.89
		B	0.95	0.92	0.96	1.30	0.84	
		C	0.94	0.75	0.80	0.89	0.87	0.82
CADMIUM	9*	A	233	456	277	271	232	266
		B	364	254	333	121	168	
		C	476	592	346	440	504	387
CHROMIUM	16.9	A	120	131	116	108	92.2	113
		B	112	116	128	121	108	
		C	140	150	126	130	119	116
LEAD	111.3	A	9890	21346	11590	13386	15461	13524
		B	10536	16815	10774	15019	15838	
		C	17267	26151	18971	19253	21392	15252
NICKEL	31.9	A	72.00	66.38	68.75	70.30	63.72	68.32
		B	68.18	69.90	71.70	75.30	66.67	
		C	72.28	67.92	70.87	70.75	64.70	67.96
MERCURY	261.5	A	10.7	18.0	12.1	10.8	18.0	8.3
		B	13.8	11.5	9.7	8.4	16.9	
		C	10.4	14.5	13.7	8.2	10.3	8.2
CHROME +6	1.6	NOT	DETECTED	BY	WESTON			

* Estimated

TABLE 8

SUMMARY OF RESULTS OF STATISTICAL ANALYSIS

	SOOTBLOW (1)				ESP FIELD (2)				DAY (3)			
	Non-Par ANOVA	1 Way ANOVA	2 Way ANOVA Non Soot Only	3 Way ANOVA Day 1 Day 2	Non Par ANOVA	1 Way ANOVA	2 Way ANOVA Non Soot	3 Way ANOVA Day 1 Day 2	Non Para. ANOVA	1 Way ANOVA	2 Way ANOVA Non Soot	3 Way ANOVA Day 1 Day 2
ARSENIC	NO	NO		NO	YES	YES	NO	NO	NO	NO	NO	NO
BERYLLIUM	NO	NO		NO	NO	NO	NO	NO	YES	NO	NO	NO
CADMIUM	NO	NO		YES	YES	YES	NO	YES	NO	NO	NO	YES
CHROMIUM	NO	NO		YES	NO	NO	NO	YES	NO	NO	NO	NO
LEAD	NO	NO		NO	YES	NO	YES	YES	NO	YES	YES	YES
NICKEL	NO	NO		NO	NO	NO	NO	NO	YES	YES	NO	NO
MERCURY	NO	NO		NO	NO	NO	NO	NO	NO	NO	NO	NO

(1) Do the metal concentrations vary by whether soot is being blown.
(2) Do the metal concentrations vary by field of the ESP.
(3) Do the metal concentrations vary by day which the samples were taken.

TABLE 9

ESP METALS: Predicting Stack Gas Concentration Based on Fly Ash Concentration

METAL	Fly Ash CONC. (ppm) (1)	Predicted (ug/m^3) (2) PART.	ACT. (ug/m^3)	RATIO
ARSENIC	75.6	.26	0.6	2.3
BERYLLIUM	0.85	.003	0.3	100
CADMIUM	451	1.55	9*	5.8
CHROMIUM	130.2	.45	16.9	38
LEAD	19715	67.6	111.3	1.6
MERCURY	7.2	0.025	261.5	10,460
NICKEL	69.1	0.24	31.9	133

(1) Average of 3rd Field Sootblowing and Non-Sootblowing.
(2) Based on Grain Loading of .0015 gr/dscf.

* Estimate

TABLE 10

INCINERATOR OPERATING DATA

ITEM	RUN	#1	#2	#3	#5	#6
O_2 @ Secondary Exit	1	7.0	7.2	7.0	7.6	7.2
	2	7.0	6.9	7.0	8.1	7.2
	3	7.1	7.0	6.9	7.4	8.4
	1/5/88	8.4	7.5	7.0	-	-
	1/6/88	7.4	7.5	7.0	-	-
	1/7/88	7.6	7.4	6.9	-	-
Temp. @ Secondary Exit	1	1805	1786	1799	1826	1875
	2	1795	1791	1805	1858	1886
	3	1826	1802	1873	1848	1837
	1/5/88	1725	1682	1844	-	-
	1/6/88	1796	1732	1802	-	-
	1/7/88	1775	1727	1807	-	-
Temp. @ Primary Middle	1	1626	1804	1558	1301	1626
	2	1564	1622	1546	1281	1538
	3	1817	1942	1783	1373	1727
	1/5/88	1574	1399	1477	-	-
	1/6/88	1700	1571	1411	-	-
	1/7/88	1795	1607	1729	-	-
Draft in Primary	1	-.21	-.17	-.14	-.19	-.14
	2	-.20	-.17	-.12	-.20	-.17
	3	-.21	-.19	-.16	-.19	-.20
	1/5/88	-.26	-.24	-.03 (30%pos)	-	-
	1/6/88	-.23	-.25	-.05 (20%pos)	-	-
	1/7/88	-.24	-.25	-.02 (30%pos)		

TABLE 11

BOILER OPERATING DATA

ITEM	RUN	#1	#2	#3	#4
Steamflow (k#/hr)	1	51.5	46.1	28.8	32.9
	2	48.0	43.6	29.4	32.0
	3	52.2	46.6	24.8	25.8
	1/5/88	48.3	47.2	-	-
	1/6/88	46.7	48.2	-	-
	1/7/88	45.5	43.2	-	-
Steam Pressure (PSI)	1	589	605	597	602
	2	589	605	596	601
	3	580	606	598	602
	1/5/88	578	597	-	-
	1/6/88	578	597	-	-
	1/7/88	532	550	-	-
Steam Temperature	1	716	636	711	671
	2	716	643	713	668
	3	716	631	716	679
	1/5/88	712	581	-	-
	1/6/88	711	577	-	-
	1/7/88	704	566	-	-
ESP Inlet Temp.	1	435	452	413	429
	2	432	435	411	432
	3	447	451	402	411
	1/5/88	428	437	-	-
	1/6/88	430	443	-	-
	1/7/88	439	437	-	-

TABLE 12

Phase I and Phase II Data vs. Throughput

ITEM		DESIGN	THROUGHPUT	PHASE I	PHASE II unit 2 only
Stack Gas Flow (acfm)	1	17,200	26,300	27,200	
	2	17,200	26,300	28,500	29,300
	3	11,400	21,000	20,700	
	4	11,400	17,300	15,100	
TOTAL		78,400	91,100	91,500	-
lbs/hr MSW & RDF		50,000	58,700	46,000	28,460*
acf/PMSW & RDF		89.3	93.1	117.8	
lbs STM/hr	1	51,500	47,300	50,600	55,500
	2	51,500	47,650	45,100	46,200
	3	34,300	31,450	27,700	-
	4	34,300	35,300	30,200	-
TOTAL		171,600	161,700	153,900	-
lbs STM/ton MSW & RDF		6,864	5,500	6,690	
% O_2(Stack)		6.7	9.9	11.3	11.2
Flow @ 6% O_2		50,500	48,000	42,000	-
Stack Temp.		400	371	373	379

* North Side Only

THE NORTH SLOPE BOROUGH'S EXPERIENCES WITH INCINERATION

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ABSTRACT

The North Slope Borough operates the largest incinerator in Alaska at Prudhoe Bay. Although the incinerator serves the oil patch industrial area, the solid waste contains a large fraction of shop and camp waste. The incinerator is a BASIC model 5000 with two cascading hearths, an auxiliary burner, twin cyclonic section afterburners, water wall construction, and a boiler section. A two-field electrostatic precipitator was retrofitted to the incinerator in 1987. Recovered heat is used to heat the solid waste plant building and an adjacent borough office/residential complex. The incinerator operates under an Air Quality Control Permit issued by the Alaska Department of Environmental Conservation.

In addition to following the Air Quality Control Regulations the permit requires the incinerator to meet a 100 ppm CO standard and a minimum combustion temperature of 1500°F. Experience with the entire operation from trash collection to flue gas monitoring indicates the difficulty of operating in the Arctic. Considerable effort and millions of dollars were invested in meeting the grain loading standard.

PRUDHOE BAY'S NEED FOR INCINERATION

PRUDHOE BAY

Prudhoe Bay, Alaska is located at the extreme northern coast of Alaska on the Arctic Ocean at a latitude of 70°15', 225 miles north of the Arctic Circle. The area is an Arctic desert, with nearly continual winds, an annual average precipitation of less than six inches and temperatures below freezing for more than 300 days per year. Prudhoe Bay, the location of the largest single oil field in North America, has become an industrial complex of oil company and service company facilities which continually develop and operate the oil field. Approximately 3,000 full time workers live and work in Prudhoe Bay, on a rotation type schedule. Their workday is generally 10 to 12 hours long since they are working in a remote location and there are no normal public or community facilities available. All the workers live in company provided camps with dormitory style housing and central dining and recreational facilities.

WILDLIFE AND PERMAFROST

The entire developed area has been established as a wildlife preserve to protect the local and migrating animal life which includes caribou, grizzly and polar bear, fox, wolves, falcon, ducks, etc. Any landfill disposal of the generated municipal wastes would have adverse impact on these animals by attracting them, resulting in danger to both the animals and the landfill operators. Polar bears do hunt humans!

Another problem with landfill disposal of municipal wastes in the Prudhoe Bay area, is that the annual average surface temperature of the ground is 25°F, making the ground continuously frozen or permafrost. Any decomposable wastes buried in the ground would never decompose because they would be permanently frozen. Digging into the permafrost is extremely difficult and very expensive; therefore, landfilling of municipal wastes is not an environmentally sound nor an economical option to the waste problem.

FACILITY HISTORY

BOROUGH POWERS

To control the problem of municipal waste disposal in the Prudhoe Bay area, the North Slope Borough (NSB), the controlling local government, established an ordinance which required all municipal wastes to be disposed of by the NSB. To accomplish this, the NSB initiated the construction of a central facility which would contain a water treatment plant, a sewage treatment plant, a shredder, and an incinerator.

FIRST INCINERATORS AND SHREDDERS

The initial facility was completed in 1979. It consisted of a Heil shredder feeding a 2000 and a 4000 pound per hour stuffer tube-type incinerators designed by R.C. Johnson. The Johnson incinerators were combined with a heat recovery boiler and a bag house. Waste heat recovered from the incinerators was used to heat the solid waste facility and the adjacent camp and office complex.

Immediately after start-up, the Heil shredder was incapable of processing the large quantities of industrial construction waste materials consisting of timbers, pallets, wooden boxes, etc. being generated in Prudhoe Bay. To resolve the problem, a Saturn 300 shredder was installed. The Saturn 300 was too small, so it was subsequently upgraded to a Saturn 600 shredder.

REPLACEMENT INCINERATOR AND SHREDDER

After two years of operation, the R.C. Johnson incinerators, boilers and bag house showed signs of severe internal corrosion. The high labor costs and the large maintenance requirements associated with the two Johnson units led the NSB to replace the two incinerators with a single mass feed Basic 5000 moving hearth incinerator in 1982. This required an expansion of the existing building to allow installation of the new Basic 5000 while the Johnson incinerators were being operated.

Since the pre-processing requirements for the Basic 5000 were different from the Johnson incinerators, a Triple/S Dynamics MSW 80 Roragator was installed. This unit met the production requirements for the Basic 5000 and provided for a smooth processing of the solid waste stream from the delivery to the final feed to the Basic 5000.

WASTE COLLECTION

COLLECTION AREA

The oil companies and service companies have facilities spread out over an area of about 500 square miles. Most of these facilities are connected with gravel roads accessing the drill sites, processing facilities, and camps. However, most exploration drilling rigs are temporarily constructed on snow pads and accessed by ice roads, a road built across the tundra, lakes or the Arctic Ocean by pumping water on the surface and allowing it to freeze.

DUMPSTERS

The NSB performs the solid waste collection by providing the waste generators with either a 27 cubic yard or a 4.7 cubic yard dumpster at a location designated by the customer. These dumpsters are picked up either on a regularly-scheduled basis or on a call-out basis. During most of the year, two full time truck drivers pick up the 27 cubic yard dumpsters using White Expeditor 240 solid waste trucks. The small 4.7 cubic yard bins are emptied on-site using a Dempster Dinosaur bin packer also mounted on a White Expeditor 240. When the quantity of waste being generated is in excess of what the two drivers can handle in a single day, a third truck and driver is added to collection effort. This normally occurs during the summer months when construction activity increases.

COLLECTION FEES

The cost of collection varies depending on the type of service provided and the location of the facility. These rates are established by the local utility board and are approved by the NSB Assembly but are not enough to pay the entire disposal costs associated with the wastes due to the extremely high cost of the facility operation. The disposal rates are shown in Table 1.

TABLE 1. DISPOSAL RATE SCHEDULE

TYPE OF SERVICE	COST PER BIN
27 Yard Bin - Scheduled	$ 825.00
27 Yard Bin - Unscheduled	1,097.25
4.7 Yard Bin - Scheduled	175.00
4.7 Yard Bin - Unscheduled	232.75
Truck Time per hour*	100.00

*Truck time is charged on pick-ups which are greater than 12 miles from the NSB Facility and charged to the nearest one half-hour.

WASTE CHARACTERIZATION

SOURCES

The solid waste collected is generated by three main types of facilities. These are drill rigs, maintenance shops and production facilities, and camp and offices. The waste stream generated by each of these types of facilities has its own characteristics associated with the type of activity at that location. The two major oil companies generate 80% of the total waste.

Drill Rigs

Drill rig wastes generally consist of shipping pallets, empty drilling mud boxes, empty cement bags, plastic wrap, oil spill clean-up absorbent pads and timbers. These wastes will always need to be shredded prior to incineration since the material is too large to fit into the loader conveyor feeding the Basic 5000.

Shops and Production Facilities

The maintenance shops and production facilities have very diverse waste generated from construction, maintenance and operations activities. Typically, this waste contains shipping pallets, oily rags and absorbents, shipping boxes and plastic wrapping, vehicle tires, rubber hoses, insulation, cut lumber, office trash, snack bar wastes, etc. This waste usually needs to be shredded prior to incineration.

Camps and Offices

Solid waste received from the camps and offices rarely contains anything but large quantities of wet material generated by the kitchens and large quantities of paper products generated by the daily cleaning of the living quarters and the office complexes.

WASTE VOLUMES

Table 2 summarizes the percentage of wastes received from the three types of facilities in 1988.

TABLE 2. WASTE VOLUMES

WASTE GENERATOR	TONS RECEIVED	PERCENT CONTRIBUTION
Drill Rigs	2,800	22%
Shops and Production Facilities	6,090	48%
Camps, Offices & Kitchens	3,800	30%
Totals	12,690	100%

COMPOSITION

Studies performed on the different type of wastes generated by these three different facilities show the moisture content varies from 37% for the wastes from the drill rigs to over 50% from the kitchens and camps. Table 3 presents the moisture content and heat and ash content data. The moisture content data does not reflect the presence of snow in the collection boxes which accumulates in significant quantities during the severe winter wind storms common to the entire North Slope of Alaska. These winter "blows" fill the collection bins with large quantities of windblown snow which is nearly impossible to separate out because the generators usually dump additional wastes on top of the accumulated snow. Separation during the pre-incineration processing is difficult because the plant and bunker area is usually at or below 32°F and significant melting does not occur. For this reason, the moisture content varies widely between the four summer months and the eight months of winter.

The amount of ash produced by the three different waste streams also varies. The kitchen and camp wastes reduce to about 7% ash when the feed rates are high and there is plenty of woody material to completely dry out the material so complete combustion can be accomplished. The heat content (Btu/lb, dry) is higher than typical municipal wastes. This is probably due to the large amount of wood and plastic wastes.

TABLE 3. WASTE CHARACTERIZATION

WASTE GENERATOR	MOISTURE CONTENT	HEAT CONTENT	ASH CONTENT
Drill Rigs	37%	7900 Btu/lb	18%
Shops and Production Facilities	42%	9000	13%
Camps, Offices & Kitchens	51%	9300	7%

WASTE STAGING

The layout of the NSB utilities building is shown in Figure 1. The 27 cubic yard waste bins are brought into one of two doors. Kitchen and camp waste bins are brought in through the west door and dumped directly onto the loading floor. Other wastes from the drilling rigs and shops are brought in through the east door and dumped directly onto the loading apron that is tilted to a conveyor belt leading to a Triple/S Dynamics shredder.

A chain saw is used to cut long timbers to size before they are conveyed to the shredder. A hydraulic boom grapple is mounted at the base of the conveyor to assist in the unloading and feeding operations, and to lift out metal objects such as pipe, fittings, or structural steel which would damage the shredder blades. The metal objects are taken directly to the ash box for disposal.

Kitchen wastes, shredder output, and wastewater sludge filter cake are mixed as they are pushed with a Caterpillar 920 front-end loader onto the incinerator feed belt. A smaller "Bobcat" front-end loader used for many years was too small. Proper mixing assures better combustion because dry wood helps to burn wet frozen garbage.

INCINERATOR CONFIGURATION AND OPERATION

BASIC 5000

The Basic 5000 incinerator was designed to handle up to 40 MM Btu/hr of municipal solid waste - up to 100 Ton/Day depending upon heat content, or 5000 lb/hr at 8000 Btu/lb. The 40 MM Btu/hr maximum input is the combined value of the waste input and the heat input from oil burners. This is one of the first Basic 5000 units built.

Figure 2 is the incinerator elevation drawing. The four-foot wide by 50 foot long conveyor belt feeds the blended waste into a 4x4x4 foot feed hopper for the incinerator. The hoppers far wall is a ram face. When the hopper is full, as indicated by a photocell, a sliding door closes the hopper's top and a gear driven ram pushes the waste through a hinged wall section door into the incinerator.

FLOOR GRATES

The incinerator's grates are two tilted, longitudinal trough-shaped hearths. The incinerator is rectangular in shape. The main chamber is approximately 9 foot wide by 12 foot high by 28 foot long. The hearths which are slightly sloped toward the outlet (ash) end are pulsed with air bladders which rapidly inflate and push the hearth into a rubber bumper causing the burning waste to cascade toward the outlet with each pulse. The pulses are set to occur about once every 20 to 100 seconds, depending upon burning characteristics. Waste residence time on the hearths is 2 to 2 1/2 hours. The incinerator's main chamber has water tube walls for heat recovery.

MAIN CHAMBER

Combustion air enters the main chamber as underfire air through air jets located along the length of the hearth at various elevations on the stepped sides (trough) of the hearth (see Figure 3, hearth details). Air also enters the chamber as leakage around the inlet ram gate and around seals between the pulse hearths and the chamber walls. The main chamber was intended to operate without excess air for combustion but that has not been possible in practice. Gas temperatures near the top of the main chamber can be below 1000°F when burning wet garbage without the oil burners.

A 105 gallon per hour maximum rating oil fired burner is mounted on the back wall above the level of the first hearth. When operating, this auxiliary burner is normally fired at 40-50 gph. During the winter the burner is cycled on and off as required to keep the combustion chamber temperature above a minimum of 1000°F. Ice, snow, frozen garbage, and air infiltration keep the combustion zone temperatures down.

Ash and char drop off the second (lower) hearth into a water bath approximately 18 inches below the end of the hearth. The submerged ash is then removed with an ash drag which dumps the ash into an ash box. Combustibles in the ash have been visually estimated at 20%, mostly during wintertime operations. The ash, along with the metals removed before the shredder, are hauled to the borough's landfill.

REBURN TUNNELS

Just above the auxiliary burner, hot flue gases exit the combustion chamber and enter two parallel horizontal 4 foot diameter cylindrical reburn tunnels where oil burners can raise the temperatures to 1700°F or higher. A 30 gph maximum oil-fired burner is mounted on the cylindrical axis of each reburn chamber. The burners can modulate from a low fire of 15 gph to a high fire of 30 gph. It was intended that all excess air be introduced into the reburn tunnels. Air for reburning is blown with four fans into the tunnels on a tangent to induce cyclonic flow.

BY-PASS STACK, BOILER AND ELECTROSTATIC PRECIPITATOR

After the reburn tunnels, the gas enters a rectangular section which is the base for a by-pass stack. After this base the hot gasses enter a high followed by a low temperature convection section. These sections are labeled "boiler" in Figure 2. A portion of the exhaust gases from the boiler section are recirculated through the boiler section to reduce the boiler inlet temperature. The recirculation fan is driven with a 40HP electric motor. Flue gases above 1150°F carry sticky flyash which adheres to the convection tubes. Originally, the tubes were finned but because of problems with fouling they were replaced with bare tubes during the electrostatic precipitator (ESP) installation. The recovered heat as hot glycol is used to heat the solid waste plant building and an adjacent borough office/residential complex. The final induced draft fan is driven with a 100 HP electric motor.

MAJOR MODIFICATIONS

Many modifications on the Basic 5000 incinerator were initiated to increase the efficiency of the incinerator. Some of the modifications were necessary to make the incinerator operate more reliably and others were to raise the internal combustion temperatures or turbulence to reduce emissions.

EFFICIENCY, EMISSIONS, AND SAFETY INSTRUMENTATION

Shortly after the start-up of the Basic 5000, the system experienced a major failure when the boiler section got too hot resulting in a rupture in the hot water piping. The safety instrumentation failed to shut down the incinerator when the boiler section reached the maximum allowable temperature. To prevent this from happening again, additional temperature and pressure sensors and automatic start-up, operation, and shut-down controls were installed.

FIRE BRICKS AND INSULATING BLANKETS

The original hearths in the Basic 5000 were lined with two layers of fire bricks. These fire bricks worked for a short time but the pulsing action of the hearths and the damaging effects of the metal objects present in the industrial wastes quickly broke out the brickwork. When one brick was lost from the hearths, the adjacent bricks quickly loosened. The wastes moving along the hearths would catch on the hole left by the lost brick and prevent the movement of the wastes through the incinerator. To solve the problem, the fire bricks were removed and the entire hearth refractory was reconstructed using a abrasion-resistant castable refractory. The configuration of the hearths was kept the same utilizing three steps on each side of the hearth to provide turning of the wastes and underfire air inlets.

The inside of the Basic 5000 main chamber was originally constructed with the boiler water tubes partially exposed to allow both radiant and convective heat transfer. Difficulties in maintaining high combustion zone temperatures were partially attributed to the heat loss to the boiler tubes and through the main chamber walls. An effort to reduce this heat loss and increase the combustion chamber temperatures included the installation of an insulating blanket on the inside walls and ceiling of the main chamber. The first installation used 1800°F materials which succeeded in increasing the combustion zone temperatures significantly, but also increased those temperatures above the operating temperature of the blanket material resulting in accelerated degradation of the blanket material. When the 1800°F blanket material deteriorated beyond its useful purpose, the blanket was removed and replaced with a 2300°F blanket material.

The 2300°F blanket held up to the higher combustion zone temperatures but in time began to pull off away from the anchoring pins. Flyash carried by the turbulence in the chamber collected in the insulating blanket material causing it to become heavier. The internal strength of the blanket material was not enough to support the weight of the accumulated flyash and the blanket ripped from the anchoring pins. At this point, the turbulence carried more flyash on top of the blanket weighing it down even more causing large sections of the blanket to pull away from the walls and ceiling.

Boiler water temperatures obtained during the operation with the blanket material indicated that sufficient heat could be extracted by the convection section and that the added capability of the radiant section was not necessary. This prompted the replacement of the main chamber blanket with a sprayed-on refractory which finally provided a long term solution for maintaining hotter combustion zone temperatures.

PULSING HEARTH MODIFICATIONS

Several modifications to the pulsing hearths were required to increase their reliability. Due to the jerking nature of the pulsing action, the hearths are continually subjected to high stresses which caused failure of the weaker components. Modifications of the yoke and bumper systems were necessary to increase the strength of these components and to provide good pulsing action.

The hearth suspension system, which allows the hearths to swing forward during the pulsing action, went through three different configurations. Originally, the hearths were suspended by linkages attached by pivot pins. These linkages were prone to cracking and required constant maintenance and replacement. They were replaced by several coils of 3/8" diameter wire rope which was wrapped around a curved support welded to the incinerator support structure and the hearth. These coils experienced chaffing and wearing to the point that they needed replacing on a regular basis and eventually were replaced by a single braided loop of 1 and 1/4" diameter wire rope. This configuration has performed very well and is still in operation.

INSTALLATION AND REMOVAL OF EXCITERS

To increase the effectiveness of the reburn tunnels, an exciter was installed in each of the two reburn tunnels. These exciters were 16 inch diameter cylindrical inserts placed in the center of the tunnels to cause a vortex flow and to inject oxygen to enhance the burning of the flue gasses. The performance of the exciters could never really be verified but their existence significantly increased the difficulty of cleaning out the accumulated flyash in the reburn tunnels.

After the installation of an electrostatic precipitator, the exciters were removed since the emission standards were being met and the reburn tunnels were no longer being operated as a final combustion zone.

INSTALLATION OF THE ELECTROSTATIC PRECIPITATOR

The State of Alaska Department of Environmental Conservation issued a Compliance Order for the correction of the emissions problem associated with the Basic 5000 incinerator. This initiated a study by the NSB to determine what modifications could be made to the Basic 5000 to bring it's emissions within the required standards. It was determined that an electrostatic precipitator would be the best option, since the NSB had poor experience with the bag houses installed on the early Johnson incinerators.

Consultants hired to study the problem and investigate the different types of precipitators available recommended that the United McGill Corporation be hired to design and construct a precipitator for the Basic 5000. United McGill installed two of their Model 150 modular precipitators connected in series to provide the necessary particulate collection. The precipitators use needled edges on the positive plates to provide the corona and flat collector plates for dust collection. The United McGill precipitator fit nicely into the existing building. Limited exterior real estate and the desire to keep the precipitator inside for maintenance purposes made this a major design consideration.

Operation of the precipitator has resulted in bringing the particulate emissions down to a level well below the required maximum limits and has provided good reliability and low maintenance.

INSTALLATION OF A NEW BOILER

The initial design of the NSB incinerator system included a Deltak heat exchanger which extracted heat from the flue gasses after they passed through the reburn tunnels and bypass exhaust stack. This boiler was constructed with finned water tubes and compressed air soot blowers. Operating and maintenance experiences indicated that there was a significant buildup of flyash on the fins of the boiler tubes which resulted in a decrease in the heat transfer efficiency of the entire boiler. The compressed air soot blowers were installed in the center of the tube stacks and did not effectively blow the fins clean. The boiler had no easy access for cleaning by the maintenance crew. It was felt that the Deltak boiler should be replaced with a new boiler which would not foul and provide fewer maintenance problems.

As a part of their contract to install an electrostatic precipitator, United Mcgill was also contracted to install a new boiler section. The design of the new boiler incorporated bare water tubes, the placement of the compressed air soot blowers over the top of the boiler tubes, the installation of sonic soot blowers in the top of the boiler, and the addition of collection hoppers under the tubed sections. This new boiler section has performed very well and the tubes are staying fairly clean. However, the heat transfer rate in the new boiler is not as good as it was with the finned tube boiler when the fins were clean and free from flyash buildup.

INSTALLATION OF A CO MONITOR

The fourth permit to operate required the NSB to monitor and report on the carbon monoxide emissions. To accomplish this, a carbon monoxide monitor was installed on the exhaust stack just above the existing opacity monitor. Both the CO monitor and the opacity monitor are installed below the roof line inside the solid waste plant to allow environmental control for the instrumentation and for ease of access for maintenance. A Dynatron CO monitor was purchased because the existing opacity monitor is also a Dynatron and there is some commonality between the parts used on both instruments providing standardized maintenance and instrument operation. Also, no calibration gas is required with the Dynatron CO monitor.

FUTURE MODIFICATIONS

WINTER OPERATION

Operational conditions during the winter months often result in combustion temperatures lower than 1500°F due to the frozen wastes, the presence of snow in the collection bins, and the lower volumes produced by the Prudhoe Bay customers. In order to increase the main chamber combustion temperatures and increase the performance of the Basic 5000, a major rework of the hearth and under-fire air system will be accomplished in 1989. This work will include complete replacement of the hearth refractory, the installation of fewer and smaller under-fire air jets, the installation of new water seals along the sides of the pulsing hearths, the replacement of the hearth end seals, and a general sealing of air leaks around the combustion chamber. It is anticipated that this work will significantly reduce the air infiltration (excess air), and provide the proper amount of under-fire air needed for more efficient combustion.

WASTE OIL AND NATURAL GAS

The NSB has investigated the possibility of collecting and incinerating used oil generated by the large fleet of vehicles and construction equipment operating in the Prudhoe Bay oil patch. Firing used oil in the auxiliary burner would maintain minimum combustion zone temperatures and to also help keep the used oil disposal problem in the region to a minimum. A feasibility study was completed for this conversion and an EPA identification number for hazardous waste activity was acquired.

An investigation into the quantity of used oil generated in the Prudhoe Bay area and the availability of the used oil for delivery to the NSB incinerator facility indicated that there were large quantities of used oil produced. But, the study also showed that almost all of the used oil was burned by the generators in their own oil burning building heaters. High heating costs and high cost of transporting the used oil to a licensed disposal site made on-site burning a most effective solution. The amounts of oil used expected to be delivered to the NSB incinerator were quite small and did not justify the conversion costs of the equipment.

Although Prudhoe Bay sits atop a huge natural gas reserve, only the oil companies are using natural gas for heating and energy production. The NSB solid waste facility and adjoining camp do not have natural gas at this time. There is, however, a company in the early stages of installing a natural gas utility which will provide the NSB facilities with less expensive natural gas. The NSB intends to convert the oil fired burners on the Basic 5000 to burn either natural gas or #2 diesel fuel oil.

LABOR REQUIREMENTS

The labor requirements for operating the incinerator vary according to the time of year. Wintertime operation usually requires a minimum of three operators; one in the control room, one on the Basic 5000 itself, and one operating the shredder and working the bunker. Since the incinerator is operated 24 hours a day, two shifts of operators are required. During the summer months, deliveries are higher which requires an increase in the staffing levels. Usually one operator is added to the shredder/bunker operation and one to keep up with normal housekeeping and assist the others in clearing up any bottlenecks in the operation. In addition to the operators, there is a full time mechanic and a full time electrician which maintain all of the NSB's facilities in Prudhoe, not just the solid waste incineration equipment.

Since the facility is located in a remote part of Alaska, the NSB's personnel work two weeks on shift in Prudhoe Bay and then get two weeks off for rest and relaxation. Therefore, staffing levels are usually about eight operators and maintenance personnel in the winter months and twelve in the summer. Of course, there are associated personnel such as the truck drivers, truck mechanics, office staff, and management involved with the operation. Wages are high and the associated indirect costs are high for all work activity necessitating rather high collection charges. The actual cost to process the waste is approximately $35 per yard including pick-up, shredding, incineration and ash disposal.

COMPLIANCE HISTORY AND PERFORMANCE TEST RESULTS

STATE AGENCY

The Alaska Department of Environmental Conservation (ADEC) is the state's environmental regulatory agency. The ADEC has been delegated the authority to enforce most provisions of the Clean Air Act. ADEC decides upon and writes all Air Quality Control Permits to Operate (AQCP), including conducting all prevention of significant deterioration (PSD) reviews.

FIRST PERMIT AND SOURCE TEST

In March 1980 the NSB made application for an AQCP for their proposed Basic 5000 incinerator. The application was denied for lack of complete source test results. In May 1980 Basic Engineering provided data to show that emissions were below PSD trigger limits (250 t/y regulated pollutant). On June 5, 1980, the ADEC issued an AQCP to the NSB for the Basic 5000. That permit required a source test within 120 days after startup to demonstrate compliance with Alaska's grain loading standard of 0.08 grains per dry standard cubic foot (gr/DSCF) at 12% CO_2 and an opacity standard of 20%. The permit expired July 30, 1983.

The Basic 5000 was initially started up in July 1981, and it was operated intermittently (as reported in April 1982) because of startup problems and problems with the firebrick.

A source test consisting of three runs conducted on July 20-24, 1982, yielded 0.47 gr/DSCF corrected to 12% CO_2. If the run with the most air leakage was thrown out, the average would then be 0.31 gr/DSCF at 12% CO_2 as listed on Table 4. All source test results are summarized in Table 4.

TABLE 4 - SOURCE TEST PARAMETERS AND RESULTS

Parameter	Test Data					
	7/82	1/84	2/85	5/85	6/85	10/87
Solid Waste lb/hr	—	~6000	—	~7600	—	~5000
Percentage Excess Air	130	98	123	130	116	152
Main Chamber:						
Temperature, °F	1500	1100	1800	1500	1600	1700
Residence, seconds	3	4	3	3.5	3.5	3.5
CO ppm	15-110	—	55	23	< 100	72
Opacity Percentage	—	0-5	14	12	7	2-3
gr/DSCF at 12% CO_2	0.31	0.074	0.32	0.14	0.12	0.038/
With Soot Blowers Off				0.088	0.069	with ESP

The auxiliary burner was known to be firing only during the 1984 and the May and June 1985 tests. The tests were conducted using EPA Methods 1-7.

Because of the poor 1982 source test results, John Basic, the original designer, evaluated the performance of the incinerator and said that he felt the incinerator was tested before it was operating properly and he made the following recommendations:

1. Increase the reburn tunnel temperatures and lower the excess air.
2. Raise the feed rate to 40 MM Btu/hr to keep operating temperatures up.

On July 6, 1983, the NSB reported that there was extensive damage in the lower hearth floor area by the ash pit and that only about 30% of the roof refractory remained and much of the insulation and tile was hanging loose.

SECOND PERMIT AND SOURCE TEST

Because the NSB was working on repairs, it was decided in July 1983 to reissue another AQCP to allow continued operation as long as the NSB continued to work toward meeting emissions standards for the Basic 5000 incinerator. The AQCP was limited to one year (July 1984) and it included an action plan for attaining compliance. In addition to a report describing modifications/improvements, the NSB was to complete the installation of a sewage sludge dewatering system by August 15, 1983, and a solid waste feed handling system by December 15, 1983. Another source test was required by January 31, 1984.

On December 14, 1983, the NSB reported that the sewage sludge press was successfully producing filter cakes.

The NSB on May 30, 1984, reported that the following recommendations were implemented after the 1982 source test but before the 1984 source test:

1. Improved sealing of combustion chambers to reduce air infiltration (cut down on excess air).
2. Installation of a bulk feed conveying system for providing proper waste input at rated capacity versus the previous shredded waste conveying system which had too many hang-up points. A new shredder was installed which produces a material size that is more compatible with the pulse hearth than was the previous material.
3. Increased the firing capacity of the oil burners on the reburn tunnels to ensure the design temperatures of 1500-1700 F are maintained.
4. Added thermal exciters in the reburn tunnels to further increase the efficiency of combustion by creating turbulence without increasing excess air. That further ensures that design temperatures are reached.
5. Installed 1/2" diameter inserts into hearth air inlet jets (previously 1" diameter) to obtain better air penetration into wastes without increasing excess air.
6. Monitoring and control of draft to limit excess air which caused suspension of flyash.
7. Improvements in the pulse hearth suspension system to predict stoking and burning of wastes to help maintain adequate combustion.
8. Installation and calibration of an opacity monitor. This allowed operators to regulate system operation to achieve lower opacities and flyash loading.
9. Provided additional operator training to explain the above improvements and emphasize operating procedures to reduce emissions.

As of March 1, 1984 the NSB reported that it had spent approximately twelve million dollars on the improvements/modifications towards attainment.

After the above modifications were completed, the Basic 5000 was source tested on January 27, 1984. The data is listed in Table 4. This is the first source test to comply with emission standards. It is also the test with the lowest percent excess air.

For all tests, the solid waste feed rate should be considered only accurate to within 30% because the compactness of the waste after shredding is not exactly known.

On August 17, 1984, the NSB informed the ADEC that the opacity meter/recorder system was operational.

THIRD PERMIT AND SOURCE TEST

On July 13, 1984, the NSB requested renewal of its AQCP. Based upon the fact that the NSB complied with the action plan in the 1983 AQCP and just passed a source test, another one year duration AQCP was issued. That new permit required operation of a continuous opacity monitor in the Basic 5000 stack. Because of the expense and poor logistics another source test was not requested. The opacity monitor would continuously verify compliance.

The NSB took it upon itself to conduct three sets of source tests in 1985. Between the February and the May 1985 tests the opacity meter was replaced with a newer model and a Kaowool insulating blanket was installed on the face of the radiant section water wall to raise the main chamber temperature. All source tests results for 1985, except one with the soot blowers off, exceeded the emission standard. On June 13, 1985, the NSB wrote the ADEC stating that they had decided to proceed with the design and construction of suitable pollution control equipment and they requested renewal of their operating permit.

COMPLIANCE ORDER, SOURCE TEST, AND FOURTH PERMIT

The ADEC responded to the NSB's request and stated that a permit could not be issued for a source violating emission standards, but suggested a compliance order by the NSB's consent as a legal instrument to allow continued operation while the NSB is retrofitting the incinerator with an emission control device. The compliance order was negotiated and signed on January 9, 1986. It required the NSB to have appropriate air pollution control equipment designed and installed by February 15, 1988, and report the source test results by May 15, 1988. Monthly operating reports were also required.

By October 1987, the NSB had a McGill Model 150 ESP installed and source tested. The McGill precipitator has two fields, and two hoppers. The incinerator with precipitator was source tested on October 13, 1987. The test results are listed in Table 4.

The opacity from the ESP is near zero except when the plates are rapped in the secondary field, or when conducting particulates are in the flue gas. Such particulates that are known to cause higher opacities are powdered graphite and soot.

On November 19, 1987, the ADEC received the test report and a renewed request for a permit to operate. On December 3, 1987, ADEC wrote back expressing concerns with some of the opacity problems in the ESP exhaust and advising the NSB to investigate possible solutions.

The ADEC received the last of the source test data on January 5, 1988. On January 13, 1988, the ADEC issued an AQCP because for the month of November 1987, the incinerator's opacity data summary indicated that the 20% opacity standard was met 99.4% of the operating time. The AQCP will expire on January 30, 1991.

New conditions were added to the permit to limit emissions of dioxins and furans. The combustion zone temperature had to be kept above 1500°F and the maximum concentration of carbon monoxide in the flue gas was to be 100ppm or less.

MANUFACTURER'S COMMENTS

Basic Environmental Engineering (BASIC) has reviewed the above discussion and has offered the following information for a better understanding of some of the problems. Most of the information which also deals with experiences at other sites is quoted from John Basic's February 20, 1989, letter.

At the time the Model 5000 was designed an analog controller was selected. These controllers were not very accurate. Control systems have since been upgraded to include more discrete digital control.

The refractory failure in the main chamber in 1983 was due to defective refractory material sold to BASIC. BASIC replaced the refractory at its own expense. Hard face refractory has been successfully applied to the water walks of a smaller Basic incinerator.

Installation of longer suspension cables, even though it reduced maintenance, has changed the lifting and tossing action of refuse on the hearths. Because of a larger arc, these longer cables require more severe forces to move the refuse along the hearth.

The exciters importance for higher thermal efficiencies in the convection section could not be noticed because of leaky main chamber seals which caused high excess air levels.

The original boiler design had two sections - a bare tube connection section followed by a finned tube economizer section. The NSB's installing contractor deleted the Deltak designed soot blowers. An economizer at another incinerator with a Deltak designed steam soot blower has never been shut down for cleaning. Steam is not available at the NSB incinerator.

The incinerator as sold did not have "turn down" features. Hence, it needs to operate at 40 MM BTU/hr input to yield temperatures necessary to achieve low CO values and good particulate burn out as demonstrated during the 1984 source test. The 1982 source tests were conducted with feed rates as a low as 28 MM BTU/hr. "That is why the unit did not pass the 1982 emission tests."

SUMMARY

Six years of operation of the Basic 5000 has necessitated several modifications to the system to get it to operate where it can meet the latest standards for emissions. At the time of this writing, the NSB is still fine-tuning the system to get it to operate in an optimum manner. Reducing the amount of excess air and using a cheaper fuel should allow the CO and temperature standards to be met at a lower operating cost. Maintenance on the system is increasing now that the incinerator is getting older and corrosion is becoming more a problem.

The incinerator's manufacturer has addressed many of the same modifications on newer incinerators operating in the 48 states. Also, because of shorter distances and better communications within the 48 states, startup and early operational problems are easier to resolve.

As most individuals who have experience with shredding operations will confess, the shredders are operated in a constant state of self-destruction. The shredder at Prudhoe Bay is no exception and is continually requiring the replacement of teeth, broken hydraulics, and other highly stressed components.

DISCLAIMER

The work described in this paper was not funded by the U.S. Environmental Protection Agency or ADEC and, therefore, the contents do not necessarily reflect the views of the Agency or the State of Alaska and no official endorsement should be inferred.

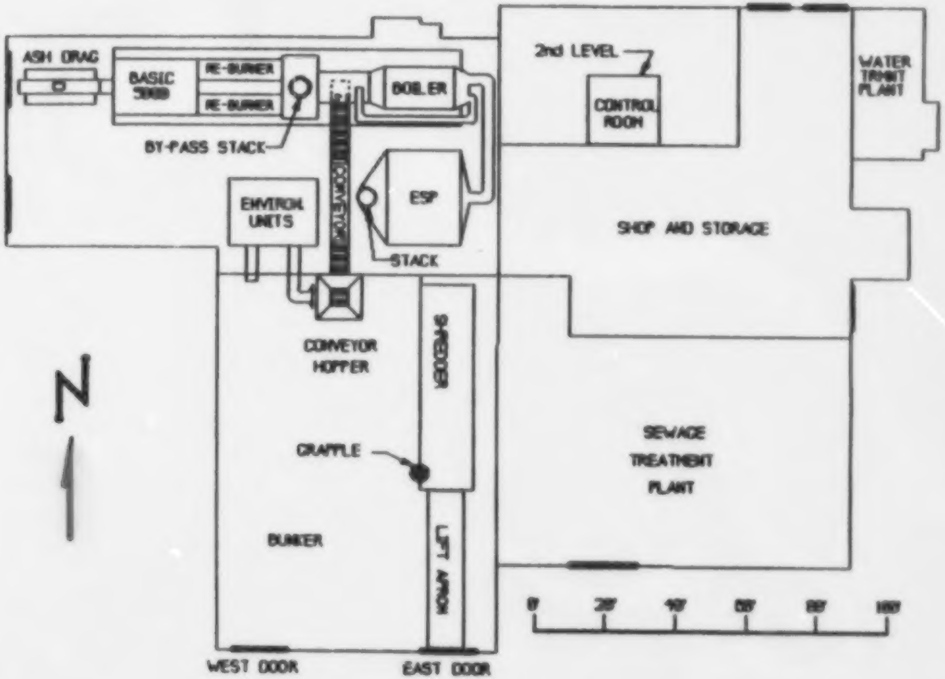
ASH DRAG
BASIC 3000
RE-BURNER
RE-BURNER
BY-PASS STACK
BOILER
CONVEYOR
ESP
STACK
ENVIRON UNITS
CONVEYOR HOPPER
SHREDDER
GRAPPLE
BUNKER
LIFT APRON
WEST DOOR
EAST DOOR
N
2nd LEVEL
CONTROL ROOM
WATER TRMNT PLANT
SHOP AND STORAGE
SEWAGE TREATMENT PLANT
0'
20'
40'
60'
80'
100'

FIGURE 1. FACILITY LAYOUT

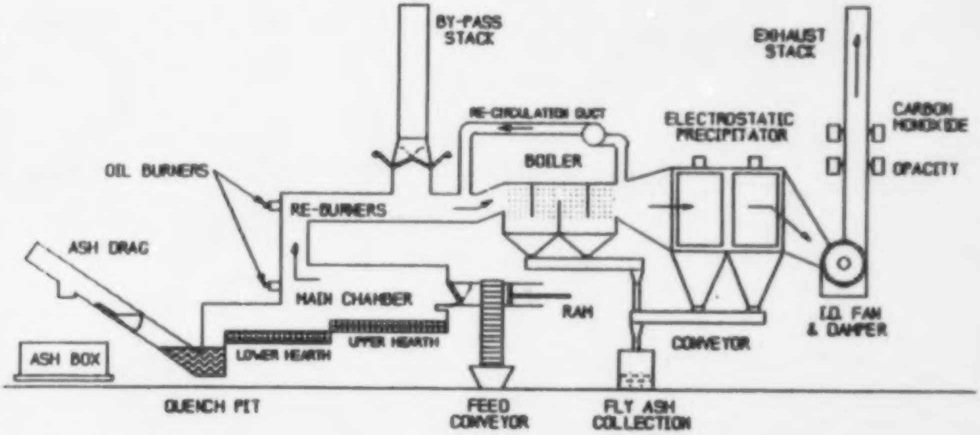
BY-PASS STACK
EXHAUST STACK
RE-CIRCULATION DUCT
ELECTROSTATIC PRECIPITATOR
CARBON MONOXIDE
OIL BURNERS
BOILER
OPACITY
RE-BURNERS
ASH DRAG
MAIN CHAMBER
RAM
I.D. FAN & DAMPER
UPPER HEARTH
CONVEYOR
ASH BOX
LOWER HEARTH
QUENCH PIT
FEED CONVEYOR
FLY ASH COLLECTION

FIGURE 2. INCINERATOR ELEVATION

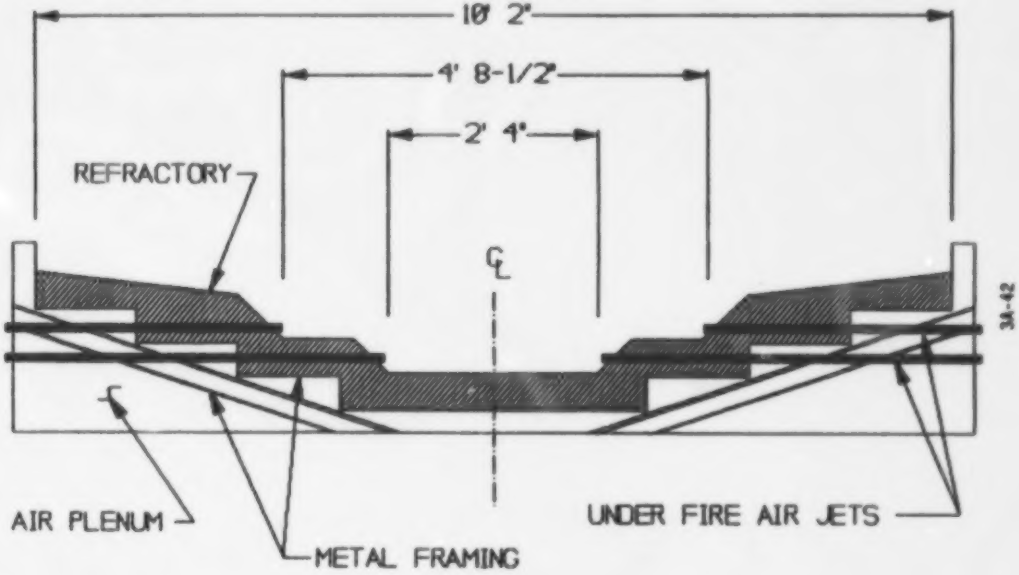
10' 2"
4' 8-1/2"
2' 4"
REFRACTORY
CL
AIR PLENUM
METAL FRAMING
UNDER FIRE AIR JETS
3A-42

FIGURE 3. INCINERATOR HEARTH

STARTUP AND OPERATIONS OF THE MID-CONNECTICUT RESOURCE RECOVERY PROJECT

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ABSTRACT

Since 1983, Combustion Engineering has made a major commitment to the waste-to-energy market. In just six years, C-E has been awarded more than $900 million in resource recovery contracts and has been one of the more active participants in new project development.

C-E subsequently had, under almost simultaneous construction schedules, three prepared fuel facilities with a combined capacity of over 8,000 tons per day (6 days per week) and over 2 million tons per year. This is equal to approximately 1 1/2 percent of the estimated municipal solid waste generated in the United States.

The 2,000 TPD Mid-Connecticut plant began startup in the fall of 1987 and achieved full scale commercial operation in the fall of 1988. The 4,000 TPD Greater Detroit facility began operation in the late fall of 1988 and, while still in startup, has processed waste in excess of the tonnage handled at Mid-Connecticut. Full commercial operation is expected by the summer of this year.

Construction of the Honolulu Resource Recovery Facility with capacity of 2,160 TPD is nearing completion and startup of that plant is expected this fall.

This paper reviews the startup and initial operations of the Mid-Connecticut project, C-E's first full scale waste-to-energy facility.

Startup test results, performance test results and operating problems encountered and solutions will be addressed.

CHARACTERIZATION OF LEACHATE FROM LANDFILLED MSWI ASH

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ABSTRACT

Data are presented on the monitoring of the quantity and quality of leachate from two municipal solid waste incinerator (MSWI) ash monofills and two MSWI ash codisposal sites, covering periods of 5 to 16 years. The data indicates that infiltration of precipitation into an ash landfill may be substantially reduced by the installation of a top layer of clay combined with a surface drainage system. Both monofill and codisposal site leachate contains high concentrations of inorganic salts and low concentrations of most trace elements. In addition, codisposal site leachate exhibits relatively high contents of e.g. organic material and ammonia, which tend to increase the potential environmental impact and complicate the leachate handling and treatment situation. A proper and consistent disposal strategy for MSWI ash must address siting and design as well as treatment and ultimate fate of the leachate in a long term perspective.

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INTRODUCTION

The development of an environmentally sound disposal strategy for municipal incinerator ash should be based on the best available knowledge of the short and long term leaching characteristics of the ash to be disposed of. The leaching behaviour of MSWI ash may be studied in the laboratory or under field conditions at existing landfills. Both laboratory and full scale studies are necessary and supplement each other. While the amount of previously reported and ongoing laboratory research on MSWI ash leaching is substantial, only limited information is available on the composition of leachate from landfilled incinerator ash, e.g. (1), (2), (3). This paper presents and discusses data on leachate production rates and leachate quality at four Danish MSWI ash landfills - two monofills and two codisposal sites.

DESCRIPTION OF DISPOSAL SITES

All four disposal sites are located in the vicinity of Copenhagen, Denmark. Two sites, 1A and 1C, are constructed as artificial ash hills near each other at one location, whereas the other two sites, 2A and 2B, are former gravel pits located within 150 m of each other at another location. Data on the design of the landfills and the amounts and types of wastes deposited are presented in table 1 and table 2.

The dominating types of waste disposed at all four disposal sites are incinerator residues (bottom ash, fly ash, and scrap metal) from a large MSWI located in the Copenhagen area. The MSWI, which is referred to as "incinerator B" in (4) and (5), consists of a moving grate boiler followed by a rotary kiln. The incinerator is equipped with electrostatic precipitators and had no acid gas scrubbers during the time in question (1973-1986). Both household waste and non-hazardous industrial waste is received at incinerator B. Descriptions and chemical analyses of bottom ash and fly ash from incinerator B have been reported in (4) and (5).

Disposal site 1A is a MSWI ash monofill which was filled up during the period 1973-1976. It holds only 10000 m^3 of incinerator residues. Like the adjacent site 1C, which holds 60000 m^3 of waste, site 1A is shaped like a rounded hill (see figure 1). Both sites are covered by a 1 m layer of topsoil with grass and other self-sown plants. Exact records of the types of material deposited at site 1C (1974-1978) do not exist, but it is known that a certain amount of incompletely combusted residues from at least one smaller incinerator as well as a portion of damaged seed corn which had been treated with mercury were deposited at site 1C along with the residues from incinerator B. Therefore, site 1C is classified as a codisposal site.

Disposal site 2A holds approximately 188500 m^3 of waste, 97% of which consists of incinerator residues, mostly from incinerator B. The remaining 3% of the waste volume consists mainly of inorganic waste. Disposal site 2A is therefore classified as a monofill. It was filled up during the period 1983-1986.

The codisposal site 2B, which holds 241000 m^3 of waste, contains a substantial amount of household waste (41000 m^3) and sludge (7000 m^3) in addition to the 182000 m^3 of incinerator residues, mostly from incinerator B. Site 2B was filled up during the period 1979-1983. Both site 2A and 2B have sloping surfaces which have been covered by a 0.5 m layer of clay overlaid by a drainage layer and a layer of topsoil (see figure 2).

More details of the design and operation of the sites 1A, 2A, and 2B have been reported in (4) and (5).

QUANTITY OF LEACHATE FROM DISPOSAL SITES

The four disposal sites are all equipped with polymer bottom liners and drainage systems for containment and collection of leachate. At each site, the leachate produced is pumped from a centrally located pumping sump and removed by tank cars at regular intervals. Records are kept of the amounts of leachate removed, and with the use of local meteorological information a water balance can be calculated for each site.

Figure 3 shows the accumulated precipitation and the accumulated amounts of leachate removed from disposal sites 1A and 1C during the period 1980-1989. The layer of topsoil does not significantly restrict the infiltration of precipitation which for both sites can be estimated at 190 mm/year, corresponding to approximately 30% of the precipitation. The "natural" water balance at site 1C was disturbed during the period 1974-1980: During 1974-1978, leachate from site 1A and another ash monofill was transferred to site 1C. The combined leachate was then pumped from the pumping sump at site 1C and sprayed into the air above the surface of site 1C during the period 1975-1978. This practice ceased when, in 1978, ash disposal was completed at site 1C, and the surface was covered by topsoil. In 1980, it was discovered that approximately 4500 m^3 of leachate had spilled over the side of the liner into the aquifer, (6). No spills or other direct disturbances have occurred at site 1C since 1980.

Figure 4 shows the accumulated precipitation and the accumulated amounts of leachate produced at disposal sites 2A and 2B during the period 1979-1988 (for site 2A 1983-1988). The effect of the top cover is clearly seen: The rate of infiltration changes from 400 mm/year (2A) and 320 mm/year (2B), corresponding to 62% and 43% of the precipitation, respectively, during the period of disposal, to 56 mm/year (2A) and 89 mm/year (2B), corresponding to 9% (2A) and 14% (2B) of the precipitation, after installation of the top cover. In 1986, it was discovered that leachate from disposal site 2B was leaking into the aquifer, (7). The estimated amount of leaked leachate has been included in the water balance calculations.

The calculated rates of infiltration are summarized in table 3.

QUALITY OF LEACHATE FROM DISPOSAL SITES

The quality of the leachate has been monitored continuously at all four disposal sites. Leachate samples have been obtained during or immediately after pumping in order to avoid sampling of stagnant leachate from the pumping sumps. The samples from sites 1A and 1C were filtered prior to analysis whereas the samples from sites 2A and 2B were analyzed unfiltered.

Because the water balances are relatively closely monitored, a L/S-value can be assigned to each sample. L/S (the liquid/solid ratio) is the ratio of the accumulated mass of leachate produced at a particular site at any time to the total mass of waste deposited at that site. This allows the composition of the leachate to be expressed as a function of L/S, which is useful for the purpose of comparing the leaching behaviour of one disposal site to another or to the results of laboratory leaching tests.

The leachate composition data are documented in the following tables:

Table 4a and 4b:	Monofill 1A	(1973-1988)
Table 5a and 5b:	Codisposal site 1C	(1980-1989)
Table 6 :	Monofill 2A	(1983-1988)
Table 7 :	Codisposal site 2B	(1979-1988)

For the sites 2A and 2B only annual averages are shown.

In table 8, the range of variation of composition of all the leachate samples from all four ash disposal sites is shown.

The analyzed leachates from MSWI ash disposal sites generally exhibit high contents of inorganic salts (primarily sodium, potassium, chloride, and sulfate) and varying levels of concentrations of other inorganic salts (e.g. calcium, carbonate/bicarbonate, ammonium/ammonia, and sulfide) as well as organic compounds (here measured as BOD5 and part of the COD). The observed concentrations of heavy metals/trace elements are generally low. pH is neutral to alkaline, and the redox potential is low enough for substantial amounts of sulfide to co-exist with sulfate.

The highest concentrations of organic material is naturally found in leachate from codisposal sites, especially during the first few years. The relatively high carbonate and ammonia concentrations in the leachate from the codisposal sites most likely originate from biodegradation of the organic material.

The moderate to high pH, the low redox potential, and the presence of carbonate and sulfide effectively control most trace element concentrations at a low level, even if they are available for leaching. This is true both for monofills and codisposal sites. However, the excessive levels of organic material, sulfide, and ammonia, which are found in some of the leachate samples from codisposal sites (or in leachate from incompletely combusted residues), become environmentally problematic and complicates treatment/discharge of the leachate. The minimal amount of organic material present in well-combusted ash appear to be sufficient to create favourable redox conditions in the saturated part of a monofill.

The concentrations of major ions are expected to decrease with time (or increasing L/S). This is seen to be true for monofill 1A (figure 5) and for codisposal site 1C, except for sodium and chloride at the latter site (figure 6). The strange behaviour of sodium chloride at site 1C is possibly caused by leaching of salt pockets which was formed by salting out of NaCl during the spraying of leachate on the surface of site 1C in 1974-1978. From time to time it has been necessary to bore holes into the side of the pumping sump to improve the flow in the bottom drain. This may have provided access to leaching of formerly hydraulicly isolated salt pockets.

Both figure 6 and figure 7, which shows the ionic strength of the leachate (calculated from the concentrations of major ions) as a function of L/S for 1A and 1C, illustrate the elevation of salt concentrations in the leachate from site 1C caused by the spraying operation. The only advantage of such an operation is a short term reduction in leachate volume. This is by far outweighed by the disadvantages in terms of increased pollution potential and operational difficulties due to scaling and formation of salt deposits.

Figure 8 illustrates the difference in organic content (BOD5) of leachate from a monofill containing well-combusted ash (site 1A) and a codisposal site (1C). This figure also indicates uneven leaching conditions with time at site 1C.

FINAL REMARKS

Land disposal of residues from municipal solid waste incinerators may pose a serious contamination risk to groundwater resources if a disposal site is located near a valuable aquifer. This is not neccessarily due to leaching of compounds normally considered hazardous but may also be caused merely by excessive leaching of soluble salts from the incinerator residues.

In order to protect aquifers (and surface water bodies) against the risk of contamination, incinerator residue disposal sites are often constructed similarly to municipal solid waste disposal sites or hazardous waste disposal sites. Usually, this implies the installation of a single or double bottom liner and a drainaige system for containment and subsequent removal of leachate. The landfill may be capped with a more or less permeable top cover designed to increase surface runoff and/or evapotranspiration and thus reduce infiltration and leachate production.

However, the direct application to municipal incinerator ash of disposal techniques which have been developed for other, completely different types of solid waste does not always comply with the inherent properties of incinerator ash, and does not, generally, lead to an overall disposal situation which is favourable in terms of minimization of environmental impact and energy consumption.

Codisposal of organic waste (e.g. household garbage) with incinerator ash is problematic, because there is a contradiction between the options available for treatment and discharge of the largely inorganic leachate from well-combusted ash and the organicly dominated leachate from household garbage. Properly sited and designed disposal alternatives which do not include the installation of bottom liners and leachate containment systems may often be advantageous, also from an environmental point of view, for well-combusted MSWI ash, (4), (5). This will seldom be the case for codisposed or incompletely combusted MSWI ash.

Landfill siting and landfill design are key elements of a proper and consistent disposal strategy which must also address the possible treatment and ultimate fate of the leachate. Failure to recognize the latter may at worst result in costly and environmentally inefficient disposal solutions.

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TABLE 1. DATA ON ASH LANDFILLS.

	SITE 1A	SITE 2A	SITE 1C	SITE 2B
Type of disposal site	Monofill	Codisposal site	Monofill	Codisposal site
Design	Artificial hill	Gravel pit	Artificial hill	Gravel pit
Bottom liner	PVC	LDPE (0.5 mm)	PVC	LDPE (0.5 mm)
Top cover	1 m topsoil	Clay + drains	1 m topsoil	Clay + drains
Total amount deposited (volume)	10000 m^3	188500 m^3	60000 m^3	241000 m^3
Total amount deposited (weight)	~10000 t	~270000 t	~60000 t	~350000 m^3
Max. depth/height	9.5 m	15 m	18.5 m	15 m
Surface area	2200 m^2	20000 m^2	8300 m^2	22900 m^2
Number of pumping sumps	1	1	1	1
Disposal period	1973-76	1983-84	1974-78	1979-83

TABLE 2. APPROXIMATE VOLUMES OF WASTE DEPOSITED AT ASH LANDFILLS.

WASTE CATEGORY	QUANTITY DEPOSITED m^3
Disposal site 1A (monofill)	
Fly ash, bottom ash and scrap metal	10000
Total	10000
Disposal site 1C (codisposal site)	
Fly ash, bottom ash, scrap metal, poorly burnt ash, Hg-treated seed corn	60000
Total	60000
Disposal site 2A (monofill)	
Fly ash, bottom ash and scrap metal	182200
Refuse from container site & recycling station	3200
Asphalt	2600
Sludge from wastewater treatment plants	400
Other	100
Total	188500
Disposal site 2B (codisposal site)	
Fly ash, bottom ash and scrap metal	182000
Refuse from container site & recycling station	11000
Sludge from wastewater treatment plants	7000
Household waste	41000
Total	241000

TABLE 3. RATE OF INFILTRATION/LEACHATE PRODUCTION RATES.

DISPOSAL SITE	RATE OF INFILTRATION BEFORE COVERING OF TOP		RATE OF INFILTRATION AFTER COVERING OF TOP	
	mm/year	% of precipitation	mm/year	% of precipitation
1A	-	-	190	30
1C	-	-	190	30
2A	400	62	56	9
2B	320	43	89	14

TABLE 4a. COMPOSITION OF LEACHATE FROM MSWI ASH MONOFILL 1A (SAMPLES FILTERED PRIOR TO ANALYSIS).

PARAMETER	UNIT	73/03/12	74/02/05	74/03/05	74/10/12	75/06/14	75/10/08	76/10/08	76/06/21	81/02/20
Temperature	deg C	30	29	29	35		21	41	24	17
TDS	mg/l	51200	34100	18500	18700	23000		13500	15300	
pH	-	10.1	8.8	9.3	8.8	8.7	9.8	9.1	9.0	10.3
Alkalinity	meq/l	3.5	2.4	2.4	1.7	4.1	5.1	2.3	1.4	1.5
Redox potential	mV		-38	-98			-108			
Conductivity	mS/m	3310	3310	2980	2700	3870	2550	3030	2560	2230
COD	mgO2/l									150
BOD5	mgO2/l									< 15
Sulfate	mg/l	3300	3700	2900	1950	3750	3000	3030	3500	4700
Sulfide	mg/l						29	51	30	2
Chloride	mg/l	11400	9800	3330	8600	9660	9400	5280	6450	5790
Total P	mg/l	0.56	0.71	0.46	0.38	0.22	0.08	0.38	0.12	0.16
Total N	mg/l		37	35	36	22	24	12	13	
Ammonia - N	mg/l	47	37	35	38	22	24	11	13	15
Na	mg/l	6000	5000	7300	4100	5575	4500	3400	4400	4300
K	mg/l	4000	4300	3750	3360	3250	2900	1790	2100	1300
Ca	mg/l	1000	615	570	500	530	440	300	520	530
Mg	mg/l	0.02	0.12	0.12	0.30	0.08	0.27	0.13	0.50	< 2
Al	mg/l	0.02	0.18	0.03	0.012	0.11	0.003	0.054	0.005	
As	mg/l	0.017	0.012	0.011	0.017	0.021	0.007	0.008	0.010	0.008
B	mg/l					0.58	0.47	0.47	0.78	
Cd	mg/l	< 0.005	< 0.001	< 0.001	< 0.005	< 0.01	< 0.005	< 0.005	0.001	< 0.0001
Co	mg/l						< 0.05	< 0.005	< 0.002	
Cr	mg/l	< 0.03	0.01		0.08	0.05	0.04	< 0.005	0.001	0.002
Cu	mg/l	0.05	< 0.001	< 0.001	< 0.005	< 0.005	0.024	0.050	0.004	0.003
Fe	mg/l	0.37	0.15	< 0.1	0.3	0.2	0.16	0.014	0.025	0.024
Hg	mg/l	< 0.00005	< 0.00005	0.0003	< 0.00005	< 0.001	[illegible]	< 0.0001	< 0.0002	0.0003
Mn	mg/l	0.3	0.04	0.03	0.03	0.03	0.05	0.03	0.02	
Ni	mg/l						< 0.05	< 0.005	< 0.005	0.002
Pb	mg/l	< 0.01	0.003	0.002	< 0.005	< 0.01	0.012	< 0.005	< 0.005	0.0029
Zn	mg/l	0.08	0.02	0.08	0.03	0.03	0.03	0.07	0.01	0.17
Leachate removed	tons	170	255	285	385	645	817	982	1076	1293
L/S	-	0.017	0.028	0.030	0.037	0.065	0.082	0.098	0.108	0.130
Ionic strength	gmol/l	0.43	0.40	0.42	0.32	0.41	0.32	0.38	0.29	0.32

TABLE 4b. COMPOSITION OF LEACHATE FROM MSWI ASH MONOFILL IA (SAMPLES FILTERED PRIOR TO ANALYSIS).

PARAMETER	UNIT	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Temperature	deg C						20	19		14
TDS	mg/l									
pH	-	10.2	9.1	9.3	9.7	9.8	10.5	10.1	10.0	10.0
Alkalinity	meq/l	2.8	1.6	1.6	2.1	2.7	4.0	4.4	4.7	2.5
Redox potential	mV	-10				-300	-200	-130	-330	-45
Conductivity	mS/m	2300	1850	2100	3300	2400	2400	1840	2740	1440
COD	mgO2/l	250	160	100	170	130	100	120	130	90
BOD5	mgO2/l	18	4	9	14	7	7	28	15	< 10
Sulfate	mg/l	2000	2300	3800	4500	4100	4400	4800	5500	3300
Sulfide	mg/l	6	1	6	8	9	6	11	20	2.4
Chloride	mg/l	5750	3700	3000	9000	6600	6600	6700	3700	2400
Total P	mg/l	0.11	0.14	0.10	0.08	0.17	0.13	0.17	0.23	0.12
Total N	mg/l	9	8	9	8	8	11	11	8	6
Ammonia - N	mg/l	7	6	7	7	6	11	9	20	6
Na	mg/l	4250	2500	4400	4200	5000	5000	6600	5400	3300
K	mg/l	1500	1300	1400	1300	1500	1470	1800	1500	890
Ca	mg/l	440	270	470	400	300	280	330	180	140
Mg	mg/l	4.9	11	13	41	5	5	4	25	2
Al	mg/l					0.017				
As	mg/l	0.009	0.018	0.020	0.035	0.025	0.036	0.035	0.037	0.025
B	mg/l									
Cd	mg/l	0.0002	< 0.0001	< 0.0001	< 0.0001		0.0005	< 0.0004	0.0037	< 0.0005
Co	mg/l									
Cr	mg/l	0.005	< 0.02	< 0.01	< 0.02		0.03	0.03	0.005	0.003
Cu	mg/l	< 0.0005	0.02	< 0.001	0.004	0.0030	0.09	0.21	0.11	0.002
Fe	mg/l	< 0.1	0.05	0.09	< 0.01	0.2	0.09	0.34	0.76	0.11
Hg	mg/l	0.0004	< 0.0002	< 0.0005	< 0.0005	< 0.0030	< 0.001	< 0.001	0.0004	0.002
Mn	mg/l									
Ni	mg/l	0.002	< 0.003	< 0.009	0.0030	0.002	< 0.01	< 0.05	< 0.05	< 0.001
Pb	mg/l	0.0013	< 0.001	< 0.0005-	0.007	0.0020	0.029	0.018	0.029	< 0.01
Zn	mg/l	0.04	< 0.02	< 0.03	< 0.02	< 0.01	0.2	0.18	0.59	0.041
Leachate removed	tons	2141	2298	2519	2843	3046	3223	3270	4087	5022
L/S	-	0.214	0.225	0.252	0.284	0.305	0.322	0.338	0.407	0.502
Ionic strength	gmol/l	0.28	0.19	0.27	0.18	0.19	0.23	0.41	0.2	0.19

TABLE 5a. COMPOSITION OF LEACHATE FROM MSWI ASH CODISPOSAL SITE 1C (SAMPLES FILTERED PRIOR TO ANALYSIS).

PARAMETER	UNIT	80/03/25	80/08/29	81/02/20	81/07/17	81/12/02	82/10/29	83/07/11
Temperature	deg C	35		> 30				
pH	-	9.5	10.1	10.3	10.0	10.5	9.9	10.7
Alkalinity	meqv/l	57	25	22	12	13	9.6	14
Redox potential	mV	-110	-160	-120	-30	-46	-65	
Conductivity	mS/m	6200	5300	5200	4900	4600	3100	2600
COD	mgO2/l		1200	1100	490	400	260	370
BOD5	mgO2/l		630	470	220	100	17	110
Sulfate	mg/l	1530	6000	4900	4500	3400	3400	2100
Sulfide	mg/l	30	22	39	25	29	12	29
Chloride	mg/l	21400	19400	17000	15100	12400	9300	11500
Total P	mg/l	4.2	0.62	0.72	0.43	0.40	0.26	0.18
Total N	mg/l	120	280	230		110	110	100
Ammonia - N	mg/l	99	280	34	170	110	110	100
Na	mg/l	13000	15000	11000	9100	8000	5600	7200
K	mg/l	8200	5300	4800	4800	4200	3400	2500
Ca	mg/l	< 2	280	410	350	420	140	220
Mg	mg/l	< 2	3.2	< 2	5.6	18	26	3.0
Al	mg/l	0.029						0.019
As	mg/l	0.034	0.0083	< 0.003	0.0045	0.002	0.005	
Cd	mg/l	0.0002	< 0.00005	< 0.0001	< 0.0001	< 0.0001	< 0.00001	
Cr	mg/l	0.009	< 0.002	0.0015	0.007	< 0.002	< 0.002	
Cu	mg/l	0.030		0.0048	0.0034	< 0.001	0.0046	
Fe	mg/l	0.23		0.070	< 0.1	0.013	< 0.01	
Hg	mg/l		0.002	0.0007	0.0003	< 0.0003	< 0.0003	
Ni	mg/l	0.020	< 0.005	0.0026	0.0035	< 0.003	0.002	
Pb	mg/l	0.004	< 0.005	0.005	< 0.0005	< 0.001	0.0002	
Zn	mg/l	< 0.1	0.10	0.16	0.030	< 0.02	< 0.02	
Leachate removed	tons	4500	4873	6700	8983	11153	14012	13211
L/S	-	0.075	0.081.	0.113	0.150	0.186	0.234	0.234
Ionic strength	gmol/l	0.75	0.83	0.68	0.59	0.52	0.39	0.45

TABLE 5b. COMPOSITION OF LEACHATE FROM MSWI ASH CODISPOSAL SITE 1C (SAMPLES FILTERED PRIOR TO ANALYSIS).

PARAMETER	UNIT	84/03/03	85/06/06	86/09/09	87/06/23	87/12/26	89/01/24
Temperature	deg C			35	26	37	3.
pH	-	10.3	10.2	10.4	10.9	10.0	9.9
Alkalinity	meqv/l	19	15	29	23	29	23.0
Redox potential	mV	-300	-290	-260	-280		
Conductivity	mS/m	2900	3600	4700		4900	
COD	mgO2/l	730	730	1100	640	520	430
BOD5	mgO2/l	300	210	660	270	210	104
Sulfate	mg/l	2800	3200	2500	3200	3000	2500
Sulfide	mg/l	67	60	160	190	264	
Chloride	mg/l	13000	12300	16100	16500	16800	18000
Total P	mg/l	0.38	0.24	0.47	0.19	0.26	0.63
Total N	mg/l	120	71	130		120	
Ammonia - N	mg/l	110	65	120	180	100	130
Na	mg/l	7700	6800	9200	10000	10900	13400
K	mg/l	3700	3600	3600	2500	3100	3300
Ca	mg/l	210	170	77	15	24	26
Mg	mg/l	4	22	11	3.3	11	
Al	mg/l	0.036					
As	mg/l	0.004		0.003	0.005	0.001	
Cd	mg/l	< 0.0001	< 0.0004	< 0.0004	< 0.0004	0.0040	< 0.0007
Cr	mg/l	< 0.001	0.011	< 0.002	0.003	0.006	
Cu	mg/l	0.015	0.002	0.08	0.19	0.013	0.033
Fe	mg/l	0.16	0.06	0.04	0.17	0.02	0.22
Hg	mg/l	< 0.001	< 0.0005	< 0.001	0.0008	< 0.001	
Ni	mg/l	0.005	< 0.01	0.007	0.008	0.023	0.029
Pb	mg/l	< 0.001	< 0.004	0.013	0.005	< 0.003	< 0.005
Zn	mg/l	< 0.01	< 0.01	0.01	0.44	0.08	0.09
Leachate removed	tons	16630	18864	20000	21232	21741	23700
L/S	-	0.278	0.314	0.325	0.326	0.352	0.393
Ionic strength	gmol/l	0.49	0.45	0.53	0.59	0.58	0.73

TABLE 6. ANNUAL AVERAGE COMPOSITION (OR RANGE OF COMPOSITION) OF LEACHATE FROM MSWI ASH MONOFILL 2A DURING THE PERIOD 1983-1988 (UNFILTERED SAMPLES). N = NUMBER OF SAMPLES.

PARAMETER	UNIT	1983 N = 4	1984 N = 4	1985 N = 4	1986 N = 4	1987 N = 3	1988 N = 4
pH	-	7.7-8.0	8.2-8.5	8.5-8.7	8.3-8.8	8.2-8.5	8.1-8.2
Alkalinity	meq/l	6.0	5.0	7.8	6.4	6.3	6.5
Conductivity	mS/m	1200	2000	3100	4400	4200	3860
COD	mgO2/l	150	200 b	310	260	360	330
BOD5	mgO2/l	29	42	83	28	29	<2-38
Sulfate	mg/l	640	1600	1100	1500	1000	740
Sulfide	mg/l	9	22	39	44	17	10 c
Chloride	mg/l	3800	9600	11000	14300	13100	13400
Total P	mg/l	0.7	0.8 a	0.9	0.6	1.1	1.4
Total N	mg/l	17	40	60	84	100	98
Na	mg/l	1600	3600 c	5000	6300	5200	5400 c
K	mg/l	1000	3900 c	4200	4200	3700	3700
Ca	mg/l	470	410	340	300	210	160 c
Mg	mg/l	46	11	10	17	7	5
Cd	mg/l	<0.0001-0.0003	<0.0004-0.002	<0.0004-0.0004	<0.0004	<0.0004-0.0008	<0.001
Co	mg/l						<0.002-0.061
Cr	mg/l	<0.0001-0.002	<0.003-0.008	0.011	0.007	0.018 a	<0.002-0.014
Fe	mg/l	0.82	0.25	0.14	0.15	0.16	0.18
Hg	mg/l					<0.0005 a	0.0012 c
Ni	mg/l	0.030				0.042 a	0.063 c
Pb	mg/l	<0.0001-0.002	<0.004-0.012	<0.004-0.022	<0.004	<0.006-0.018	<0.067
Zn	mg/l	<0.01-0.13	<0.02-0.10	0.073	<0.02-0.22	0.24	0.12
Leachate removed	tons	2370	10301	19054	27986	21530	26233
L/S	-	0.012	0.038	0.071	0.104	0.109	0.112
Ionic strength	mol/l	0.15	0.32	0.36	0.45	0.30	0.28

a: N = 1 b: N = 2 c: N = 3

TABLE 7. ANNUAL AVERAGE COMPOSITION (OR RANGE OF COMPOSITION) OF LEACHATE FROM MSWI ASH CODISPOSAL SITE [illegible] DURING THE PERIOD 1979-1988 (UNFILTERED SAMPLES). N = NUMBER OF SAMPLES.

PARAMETER	UNIT	1979 - 1980 N = 6	1981 N = 5	1982 N = 4	1983 N = 4	1984 N = 4	1985 N = 4	1986 N = 4	1987 N = 3	1988 N = 4
pH	-	6.8-7.8	7.2-8.0	7.8-8.1	8.0-8.3	8.4-8.7	7.1-8.5	7.5-7.8	7.5-7.7	7.2-7.8
Alkalinity	meq/l	77	46	42	42	32	43	54	55	53
Conductivity	mS/m	3800	3300	2700	2800	3100	3000	3200	3100	2200
COD	mgO2/l	400	2000	330	330	340	370	370	350	460
BOD7	mgO2/l	530	1300	38	27	16	24	19	14	13
Sulfate	mg/l	800	1400	1500	910	1700	660	580	800	550
Sulfide	mg/l	90	[illegible]	3.6	6.6	<0.05-54	2.1	<0.05-0.72	<0.02-0.30	<0.02-1.2
Chloride	mg/l	2400	1630	7400	6500	8500	9600	9000	8000	8740
Total P	mg/l	3.8	2.1	1.9	1.8		2.9	2.6	2.7	2.4
Total N	mg/l	150	150	170	180	220	220	310	300	280
Na	mg/l		6400	4500	4500	4900	4700	5200	4100	
K	mg/l		4700	4600	3600	3800	2500	3000	4000	
Ca	mg/l		310	130	72	35	38	150	100	165
Mg	mg/l		61	38	40	16	25	44	61	57
Cd	mg/l	0.0071	0.0035	0.0036		0.008	0.0037	<0.0004-0.002	0.0046	<0.0004-0.001
Co	mg/l									0.022
Cr	mg/l	0.038	0.014	0.037	0.037	<0.003-0.108	<0.003-0.094	0.086	(0.031)	0.040
Fe	mg/l	53	30	1.6	0.22	0.55	1.1	2.2	1.8	0.72
Hg	mg/l									0.0004
Ni	mg/l									0.039
Pb	mg/l	0.28	<0.003-0.38	<0.005-0.45	<0.001-0.100	<0.009-0.095	0.017	<0.004-0.013	<0.006-0.017	<0.007
Zn	mg/l	13	0.53	3.5	0.11	0.30	0.22	0.54	0.23	0.06
Leachate produced	tons	6300	3500	2700	2700	2800	3000	2300	3400	3700
L/S	-	0.024	0.096	0.065	0.076	0.082	0.088	0.059	0.065	0.11
Ionic strength	mol/l		0.30	0.33	0.31	0.36	0.33	0.35	0.34	

TABLE 8. RANGE OF VARIATION OF COMPOSITION OF LEACHATES FROM MSWI ASH LANDFILLS.

PARAMETER	UNIT	INCINERATOR ASH MONOFILLS		INCINERATOR ASH CODISPOSAL SITES	
		SITE 1A	SITE 2A	SITE 1C	SITE 2B
pH	-	8.7-10.5	7.7-8.7	9.5-10.7	6.2-8.7
Conductivity	mS/m	1650-3340	890-4700	2300-6300	2300-3600
TDS	mg/l	18800-51200	-	-	-
Alkalinity	meqv/l	1.4-5.1	4.0-9.0	9.6-57	10-89
Cl^-	mg/l	3700-11400	3800-15100	9300-21400	7600-22300
SO_4^{2-}	mg/l	1950-5800	610-2160	1900-6000	350-2660
N(-III)	mg/l	1-29	4-65	13-200	<0.02-630
Na^+	mg/l	3900-7300	1200-6600	5600-15000	2600-5800
K^+	mg/l	650-4300	680-5200	3100-8200	1500-4600
Ca^{2+}	mg/l	140-1000	92-600	<2-420	15-880
Mg^{2+}	mg/l	0.09-41	2-100	<2-22	13-150
Al	mg/l	0.003-0.18	-	0.019-0.036[a]	-
As	mg/l	0.006-0.025	-	0.001-0.034	-
Cd	mg/l	<0.0001-0.001	<0.00005-0.022	<0.00001-0.0005	<0.0001-0.025
Cr	mg/l	<0.002-0.08	<0.0001-0.018	<0.001-0.011	0.001-0.094
Cu	mg/l	<0.0006-0.21	-	<0.001-0.19	-
Hg	mg/l	<0.00006-0.0022	<0.0005-0.0017[b]	<0.0003-0.002	0.0008-0.0014
Pb	mg/l	<0.0005-0.029	<0.0001-0.032	<0.0005-0.013	<0.001-0.49
Zn	mg/l	<0.01-0.59	<0.01-0.69	<0.01-0.81	0.026-13
BOD_5	mgO_2/l	4-26	<2-100	17-660	11-6000
COD ($Cr_2O_7^{2-}$)	mgO_2/l	100-350	130-560	260-1300	220-12000
Total N	mg/l	6-37	17-140	71-280	38-380
L/S range (approximate)		0.00-0.50	0.00-0.11	0.06-0.60	0.00-0.11
No. of samples		18	24	13	28
Period of sampling		1973-1988	1983-1988	1980-1989	1979-1988
% fly ash (w/w)		Unknown	10-15	Unknown	~10
Filtration of samples ?		Yes	No	Yes	No

[b] Only 5 samples. [a] Only 3 samples.

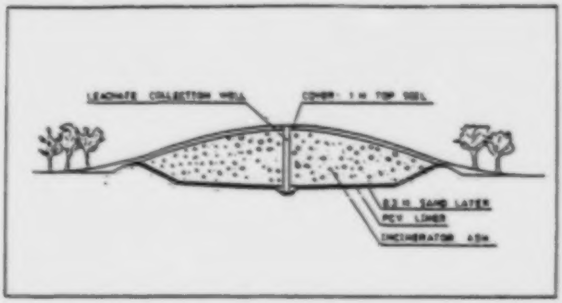
LEACHATE COLLECTION WELL
COVER: 1 M TOP SOIL
0.2 M SAND LAYER
PCV LINER
INCINERATOR ASH

Figure 1. Crossection showing principle of design of MSWI ash disposal sites 1A and 1C.

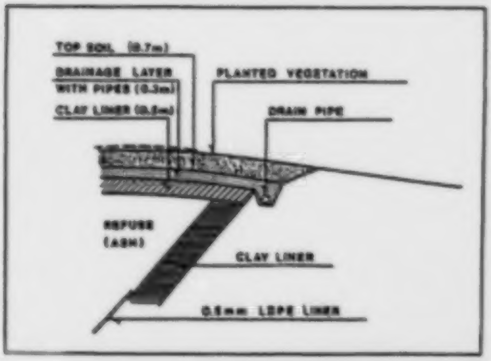
TOP SOIL (0.7m)
DRAINAGE LAYER WITH PIPES (0.3m)
CLAY LINER (0.5m)
PLANTED VEGETATION
DRAIN PIPE
REFUSE (ASH)
CLAY LINER
0.5mm LDPE LINER

Figure 2. Crossection showing principle of design of top cover of MSWI ash disposal sites 2 A and 2B.

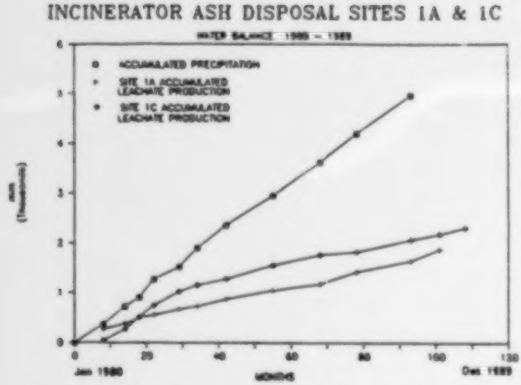

Figure 3. Accumulated precipitation and leachate production at MSWI ash sites 1A and 1C. Data on precipitation provided by the Hydrotechnical Laboratory, The Royal Veterinary and Agricultural University, Copenhagen.

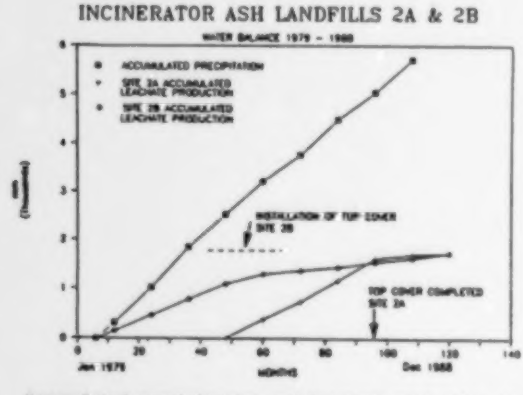

Figure 4. Accumulated precipitation and leachate production at MSWI ash sites 2A and 2B.

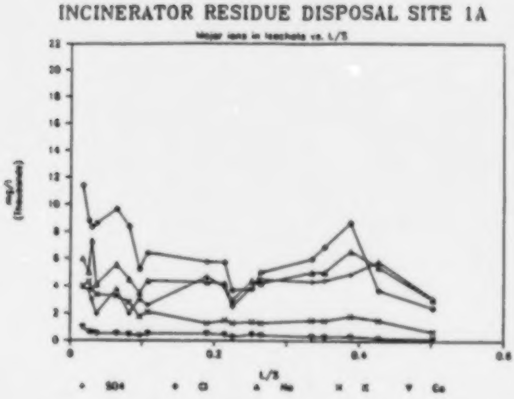

Figure 5. Major ions in leachate from MSWI ash monofill 1A vs. L/S.

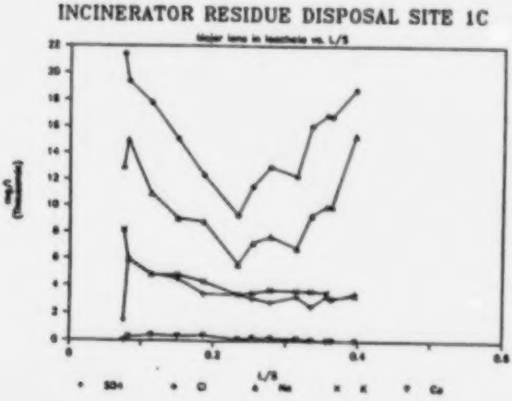

Figure 6. Major ions in leachate from MSWI ash codisposal site 1C vs. L/S.

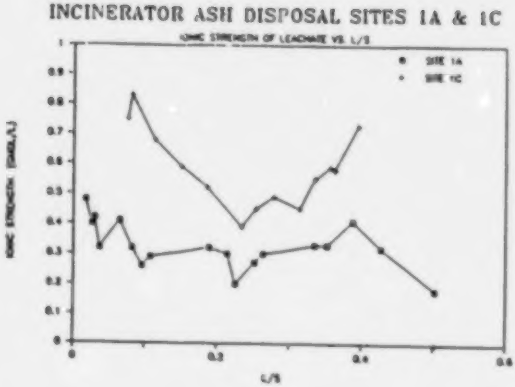

Figure 7. Ionic strength in leachate from MSWI ash disposal sites 1A and 1C vs. L/S.

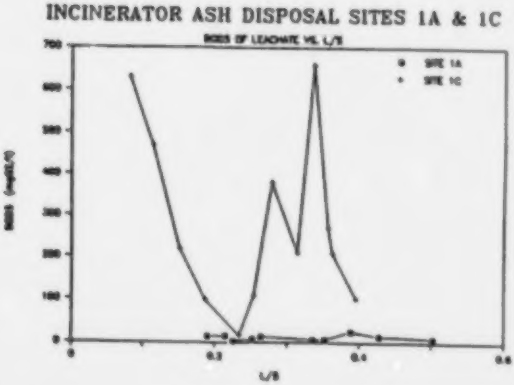

Figure 8. Biological oxygen demand of leachate from MSWI ash disposal sites 1A and 1C vs. L/S.

LEACHING PROPERTIES OF NATURAL AND STABILIZED FLUEGAS CLEANING RESIDUES FROM WASTE INCINERATION

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ABSTRACT

Waste incineration flue gas cleaning residues from four different processes were studied regarding chemical and physical qualities. Both dry and wet techniques were included. The residues were leached by the ENA-method and the concentrations of metals and salts in the leachate were analysed. The results showed considerable differences in leachate concentrations between the residues from the different processes. A series of stabilizations with additives such as cement, coal fly ash, silicate, sulphide and gypsum was made with the semi-dry residue to reduce leaching of lead. A pollution scenario model was applied to the leaching of lead from the disposal of residues from a waste incinerator with a capacity of 200,000 ton/year. Three different flue gas cleaning processes were comparedin the scenario, together with one cement stabilisation alternative.

INTRODUCTION

In Sweden about 50% - 1,300,000 tons - of household refuse is incinerated. The remaining amount is recycled or deposited. By 1992 all refuse incineration plants are to have installed advanced flue gas cleaning in which gaseous emissions such as HCl and Hg, as well as organic pollutants such as dioxins, are separated. Atmospheric pollution is decreasing, at the same time as the potential spread of pollution in the ground and water is increasing because of the expanding volume of residues produced by advanced flue gas cleaning.

At the request of the Swedish Committee for Research in Heat Technology, a number of flue gas cleaning residues have been investigated in regard to their physical, chemical and technical properties. The objective was to assess the leaching of salts and heavy metals in the deposit of residues and the measures, stabilization method and depositing technique that reduce leaching most effectively. Environmental scenarios, on the pattern of a model produced at the SGI, have been used to simulate leaching of contaminants from flue gas cleaning residues deposited in accordance with various methods.

The investigation comprised flue gas cleaning residues from four different cleaning processes:

- the semi-dry process (SD) where a lime slurry is sprayed into the flue gases and separated together with fine fly ash in a pipe filter.

- condensing flue gas cleaning (C) where the pollutants are washed out of the flue gases in a washing tower. The polluted water is neutralized, precipitated, flocculated, dewatered and mixed with the fly ash. Some of the remaining water vapour is condensed and the heat recovered.

- semi-dry flue gas cleaning followed by condensation of water vapour in the flue gases (SDC). The polluted water is purified as in condensing flue gas cleaning and the sludge mixed with the dry residue.

- separation of fly ash in a fabric filter after incineration of RDF in a fluidized limestone bed (FC).

Flue gas cleaning residues from a total of nine plants were investigated in the study. Four used the dry method, three the wet method, one the method of dry flue gas cleaning combined with condensing and one the fluidizing circulating limestone bed. In the last mentioned method, a sorted RDF consisting of 50% of the total volume of refuse is incinerated. The remaining half, consisting of metal and glass is deposited.

The size of the plants varies between 13,000 - 250,000 tons of incinerated refuse/year. The amount of flue gas cleaning residues is indicated in Table 1.

TABLE 1. Produced amount of flue gas cleaning residues per ton of incinerated refuse from advanced flue gas cleaning.

	kg/ton refuse
Semi-dry	18 - 35
Condensing	18 - 31
Semi-dry + condensing	10
Fluidized limestone bed	70

DESCRIPTION OF THE BASIC RESIDUES

Leaching caracteristics

The flue gas cleaning residues have been investigated by analysis of solid phase concentration, leaching tests with analysis of leachate concentrations and calculation of the proportions leached. Only the end products of flue gas cleaning were considered - coarse fly ash or slag separated earlier in the process is excluded.

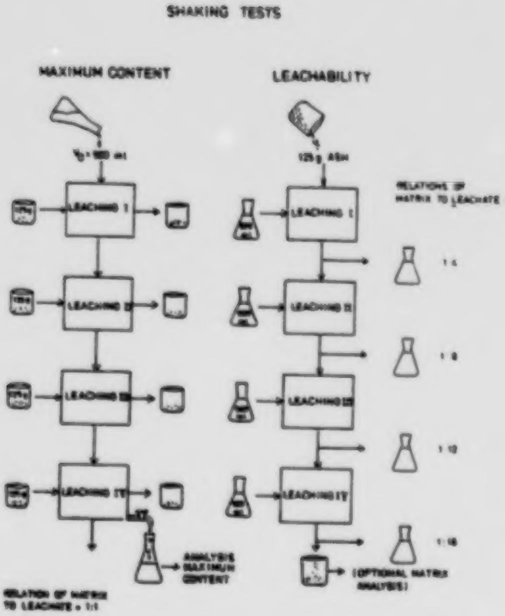
The solid content analyses for a number of elements in the residues of the various processes are documented in Table 2.

TABLE 2. Solid phase content in the flue gas cleaning residues (ppm).

	SD	SDC	FC	C
As	40-180	9.0	20	130-150
Ca	170000-290000	270000	170000	93000-104000
Cd	-	-	-	-
Co	4-11	13	13	14-25
Cr	220-570	230	270	480-660
Cu	450-860	730	4500	990-1400
Ni	20-40	40	80	90-110
Pb	4000-6300	5000	3100	5900-8100
Zn	12000-19000	14000	3800	20000
Cl	92000-22000	253000	43000	48000-56000

From the solid phase concentrations indicated, it can be seen that the variation between the residues from semi-dry, semi-dry with condensing and condensing flue gas cleaning is at most a factor of 5. This dispersion also exists between the residues from the various semi-dry plants. The fluidized limestone bed gives a residue with somewhat different solid phase concentrations. The copper concentration in particular is considerably higher, while the lead and zinc concentrations are lower. The relationship between the solid phase concentrations of different residues is interesting to bear in mind when considering the leaching data for the different residues.

The leaching of the residues has been carried out in a shaking experiment conforming to the ENA method (1). Leachate/solid (L/S) proportions of 4, 8, 12 and 16 are produced by shaking 125 g of the TS material with 500 g deionized water, pH=4, for 1 day, after which the water is separated and filtered (L/S = 4). New water is added to the material and leaching repeated. After 4 leaching cycles, i.e. 4 days, the test is terminated. L/S = 1 is obtained by leaching 125 g new material in 500 g water once a day for 4 days. The leaching cycle is described diagrammatically in Figure 1.



SHAKING TESTS
MAXIMUM CONTENT
LEACHABILITY
125 g ASH
RELATIONS OF MATRIX TO LEACHATE
LEACHING I
LEACHING II
LEACHING III
LEACHING IV
1:4
1:8
1:12
1:16
ANALYSIS MAXIMUM CONTENT
(OPTIONAL MATRIX ANALYSIS)
RELATION OF MATRIX TO LEACHATE = 1:1

Figure 1. Leaching cycle for shaking tests conforming to the ENA method (1).

The leachates obtained are analysed in regard to pH, conductivity, anions and heavy metals (ICP).

The results of the leaching tests show that the chemical composition of leachates generally varies between the residues from the various flue gas cleaning processes. Table 3 gives the range of pH, electrical conductivity and chloride ions for the various residues.

TABLE 3. pH, electric conductivity and chloride concentrations in leachate from the different residues.

	pH	κ (mS/cm)		Cl^- (g/l)	
		L/S=1	L/S=16	L/S=1	L/S=16
SD	11-13	100-170	6-10	70-140	1-30
SDC	11-13	150	16	130	5
FC	12-13	30	3	15	1
C	10, 12-13	20-80	2-7	5-35	0.1-2

pH is generally very high for leachates from all the residues, apart from two of the three plants with condensing flue gas cleaning, where the leachate had pH=10 throughout the leaching cycle. The chloride concentrations show a high initial leaching from the residues from SD and SDC, which are 2-30 times higher than the initial concentrations of the residues from C and FC. The chloride concentration in all the residues decreased almost logarithmically with increasing L/S ratio to about the same order of size as the concentration at L/S = 16.

The metal concentrations in the leachates are generally higher for the residues from SD compared with the concentrations for the residues from C and FC. This applies especially in the case of lead and zinc, where very much higher concentrations were obtained. A clear pattern of higher concentrations was also obtained for arsenic, cadmium, copper and nickel. In the residue from SDC, the leachate concentrations are comparable with those from process C, apart from lead, zinc and copper, where the concentrations correspond to those from the dry flue gas cleaning method. Figure 2 shows the ranges of concentrations for cadmium, copper, lead and zinc as a function of the L/S ratio obtained for the residue in each process.

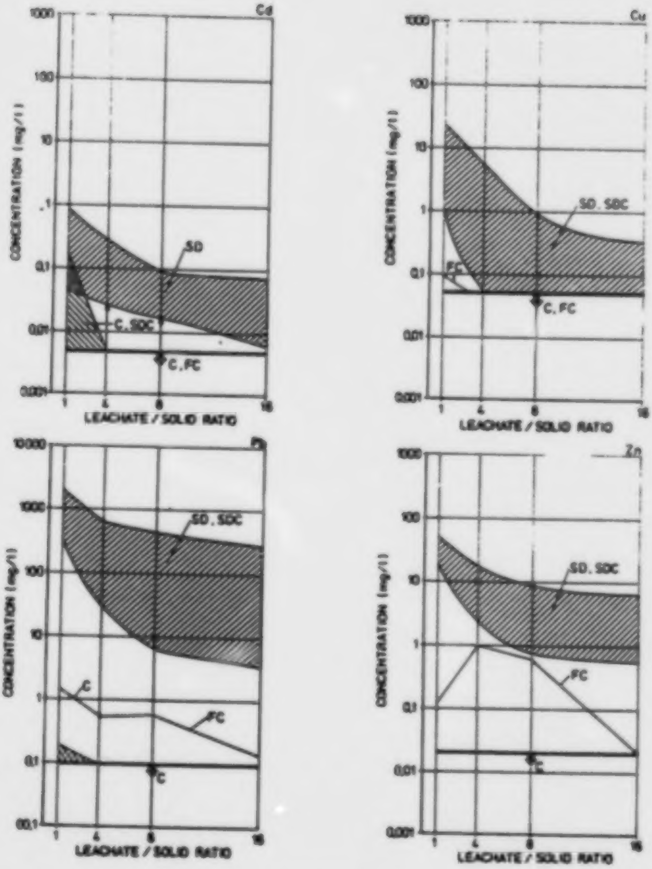


Figure 2. Ranges of leachate concentrations as a function of the L/S ratio for the different processes for Cd, Cu, Pb and Zn.

The leached proportion of the solid phase concentration constitutes a complement to the leachate concentrations in determining how easily the residue is leached. Table 4 shows the leaching up to and including L/S = 16 as a percentage of the solid content for the residues from the various processes.

TABLE 4. Leached proportions of trace elements from the solid phase for the residues from the various processes (%).

	SD	SDC	FC	C
As	<1.8 - 13	<0.5	<1.8	<0.3
Ca	>32 - 90	71	8.6	17
Cd	<0.3 - 0.7	-	-	-
Co	<9 - 31	<12	<10	<11
Cr	<0.1 - 0.9	<0.2	<0.2	<0.1
Cu	0.4 - 3.2	1.5	<0.01	<0.08
Mo	19 - 54	<7	4.7	<37
Ni	2.9 - 16	<2	<0.4	<0.3
Pb	10 - 47	28	0.2	<0.03
Zn	0.4 - 0.7	1.0	<0.2	<0.002
Cl^-	-	160	150	200

The table shows that the residues from SD and SDC are considerably more easily leached than the residues from FC and C. It is particularly noticeable that up to 47% of the lead content and up to 90% of the calcium content is leached in the leaching cycle for the residue from SD.

Physical parameters

The permeability of deposited residues is decisive for the amount of pollutants leached. Therefore, the residues should be compacted before deposition in order to reduce permeability. The residue from the semidry process has pozzolanic properties and begins to harden after moistening before it leaves the plant. Tests showed that the compaction result deteriorated the longer the residue had been stored after moistening prior to compaction. Dry residue from process SD was moistened in the laboratory to w = 19% and was stored at 7°C for 0.5 h, 4 h, 24 h, 2 days and 7 days, after which it was compacted and the dry density measured. The samples were stored at 7°C for a further 7 days, after which compressive strength and permeability were measured. The results are given in Table 5.

TABLE 5. Influence of storage on moistened residue from semi-dry flue gas cleaning with regard to compressibility, compressive strength and permeability.

Time	ρd (ton/m^3)	Water content w (%)	Compressive strength (kPa)	Saturated hydraulic conductivity k (m/s)
0.5 h	1.55	20	2200	$1 \cdot 10^{-9}$
4 h	1.47	19	1380	$1 \cdot 10^{-9}$
24 h	1.40	19	2100	$2 \cdot 10^{-9}$
2 days	1.34	19	900	$2 \cdot 10^{-8}$
7 days	1.23	19	660	$4 \cdot 10^{-8}$

The test showed that the degree of compaction decreases rapidly after the residue is moistened and that also compressive strength decreases for the compacted residue. In comparison with earlier studies at SGI (2) ageing of the mositened flue gas cleaning product shows a greater deterioration in the compaction result than if a water content of ± 10% had been applied to a fresh moistened residue (see Figure 3).

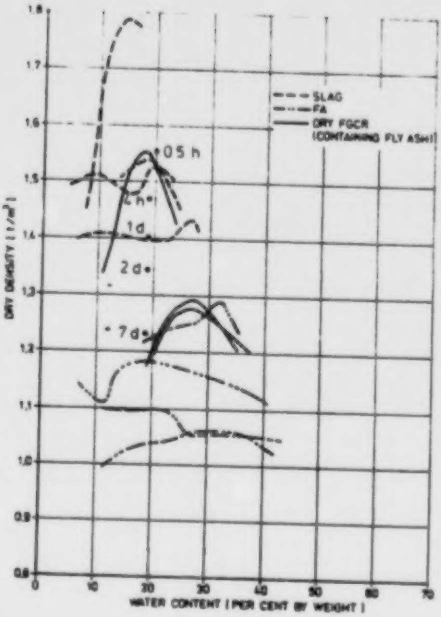


Figure 3. Compaction densities for an aged, moistened flue gas cleaning residue in comparison with earlier compaction tests made on flue gas cleaning residues (2).

The permeability of the compacted samples increases by a factor of 40 when the residue has been stored in a moist condition, see Figure 4. Permeability is higher after 7 days' storage compared to 1 day also for the loosely stored moistened semi-dry residue. A loosely stored mixed residue from condensing flue gas cleaning demonstrates a permeability of the same order of size as in loose storage of the semi-dry flue gas cleaning residue - $5 \cdot 10^{-5}$ - $1 \cdot 10^{-4}$ m/s.

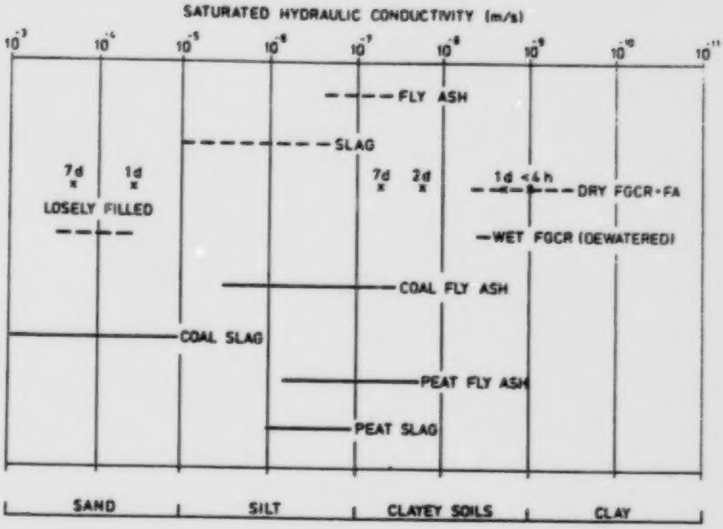


Figure 4. Saturated hydraulic conductivities of aged semi-dry residue compacted after 0.5 h, 4 h, 24 h, 2 days and 7 days, loosely filled semi-dry residue after 1 day and 7 days and loosely filled mixed condensing residue. Saturated hydraulic conductivities for compacted residues from waste incineration and coal and peat combustion are presented as a comparison (2).

STABILIZATION OF FLUE GAS CLEANING RESIDUES

Stabilization of semi-dry flue gas cleaning residue

Several leaching tests made on the semi-dry flue gas cleaning residue record the concentrations of L/S=1 for lead in a leachate of 500-2000 mg/l. In this project, a patent search has been carried out to find stabilization materials and techniques that can bind the lead in the residue. From this search, it has been judged that the following stabilizers are of interest for testing:

- Cement (Standard Portland)
- Calcium sulphate (pure gypsum)
- Waste gypsum from phosphoric acid manufacture
- Bed material from PFBC
- Cyclone ash from PFBC
- Vermiculite (an expanded silicate material)
- Sodium metasilicate
- Bentonite
- Sodium sulphide

An extensive series of tests has been performed with these stabilizers, added in various proportions. However, the flue gas cleaning residue always constituted at least 50% of the dry mix.

Each stabilizer and flue gas cleaning residue was first mixed dry. Water was then added to produce a consistency similar to a thick slurry. The various stabilizers require different amounts of water to produce the same consistency. Therefore it has not been possible to use a constant water content. The semi-dry product is strongly reactive and generates considerable heat during mixing.

A total of 35 stabilizers were produced. All of these hardened. Single-axis compressive tests after 28 days showed a strength of 700-2 000 kPa, in one case 12 000 kPa (50% cement mix).

Leaching tests were conducted on all the stabilized probes in accordance with a Uniform Leach Procedure (ULP) (3). In this test, a sample is leached statically in water (pH=4 at the start of the test) normally for 14 days. In this study, leaching continued for 128 days. The water was exchanged at regular intervals so as to obtain the maximal gradient during the test. For each basic element, leaching is recorded in the form of an effective diffusion constant (m^2/s) in those cases where the leachable fraction is known, or as a specific leaching ($kg/m^2 \cdot day$).

Despite a relatively satisfactory compressive strength, a number of the samples collapsed during the leaching. This occurred with samples stabilized with sodium sulphide, absol and waste gypsum from phosphoric acid manufacture. The best physical and chemical stabilization effect was obtained with cement, vermiculite, bentonite and sodium metasilicate. A good stabilization effect can be obtained by adding water alone in relatively large amounts. Figure 5 shows the specific leaching during the first 28 days of the tests using stabilization with water alone and with cement. Test data from cement stabilization (80% SD + 20% cement) have been used for calculating the environmental scenario later on.

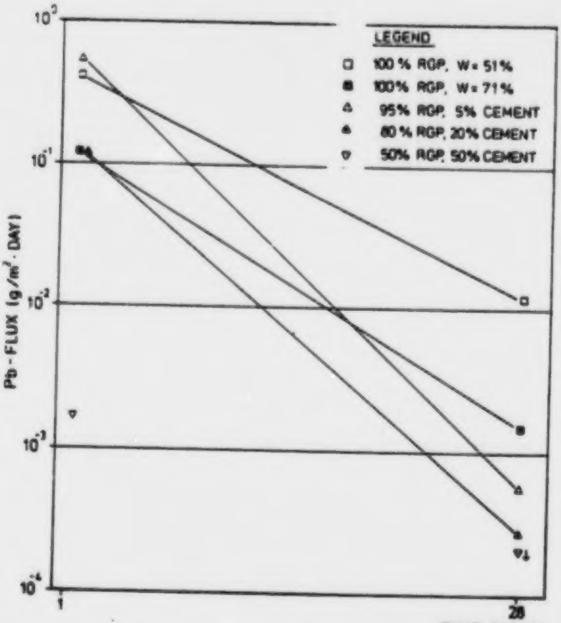


Figure 5. Specific leaching of lead in ULP surface leaching tests on stabilized samples based on semi-dry flue gas cleaning residue with the addition of water and cement.

Stabilization of the condensing residue

In the wet flue gas cleaning process, two residues are produced that are normally mixed together. In leaching of fly ash alone, very high leachate concentrations of cadmium, lead and zinc, among other elements, occur. Mixing fly ash and sludge leads to a stabilizing effect where the leachate concentration from the leaching of the mixture is lower than that produced by a combination of the leachate concentrations from each residue. Table 6 contains an example of the effect of this mix on the leachate concentrations of cadmium, lead and zinc.

TABLE 6. Leachate concentrations for fly ash, sludge and a mixture of fly ash and sludge from the wet flue gas cleaning method with L/S = 1 (L/S = 4 for sludge) (mg/l).

	Fly ash	Sludge	Mix 13% TS sludge - 87% TS fly ash
Cd	23,1	<0,005	<0,005
Pb	9,7	0,09	<0,10
Zn	161	0,072	<0,02

DEPOSITION OF FLUE GAS CLEANING RESIDUES

In Sweden, deposition of flue gas cleaning residues from refuse incineration takes place in accordance with general rules. Slag and flue gas cleaning residues together with fly ash have to be deposited separately and covered with a sealing layer of low permeability. The average net infiltration in natural ground in Sweden is about 400 mm/year. The permeability of the sealing must limit infiltration to 50 mm/year according to the ENA study. The pollution occurring as a result of the percolating leachates must be tolerated by the ecological system since there are no requirements on bottom sealing. Refuse sites must therefore be placed in areas where satisfactory dilution of the concentrated leachate can take place. Pollution of the local environment will thus vary widely depending on the dilution.

At the same time as atmospheric emissions from refuse incineration plants decrease, the requirements on environmentally satisfactory deposits will increase, since the the amount of residue produced per ton of refuse varies between the processes, as does leachability. The environmental scenario model, which has been developed by SGI (4), is based on percolation through the residue, the leaching characteristics and transfer of the leaching result to the particular object. The relation Leachate/Solid, L/S=1, 4, 8, 12 and 16 is transferred so as to apply to each point in time for the corresponding accumulated leachate production/amount of residue in the deposit. For stabilized residues the specific leaching from the ULP-test is applied. The model gives both the accumulated leaching and the concentration as a function of time for the simulated substance.

The environmental scenario model has been applied to the residues from the processes investigated. Data from one plant representing each process has been used. The semi-dry method with condensing has been excluded from the calculations owing to incomplete data. The calculations have been based on production of flue gas cleaning residues from a refuse incineration plant with a capacity of 200 000 tons of refuse per year. In Sweden, this corresponds to the amount produced by some 500 000 people except for FC where the corresponding amount of people is 1 000 000. Production of flue gas cleaning residues, including fly ash, varies between 11-70 kg/ton incinerated refuse, depending on the process. The total production of residues from the incineration of 200 000 tons of household refuse for the various processes is:

- 6930 tons for the semi-dry process
- 13 980 tons for the fluidized limestone bed process
- 2 140 tons for condensing flue gas cleaning

In addition, a deposition alternative with cement stabilization of the semi-dry product has been performed with 25% cement, giving 8 660 tons of stabilized residue.

In these cases, the calculations analyse the pollution caused by refuse incineration via the residues, i.e. fly ash together with an flue gas cleaning residue. Atmospheric emissions are excluded. As shown by the chemical analysis, leaching of lead is high in the case of the semi-dry residue. The stabilization described has also focussed on leaching of lead. The pollution calculation shown here thus relates to leaching of lead and to the applicable Swedish norms.

The following assumptions apply to the pollution calculations:

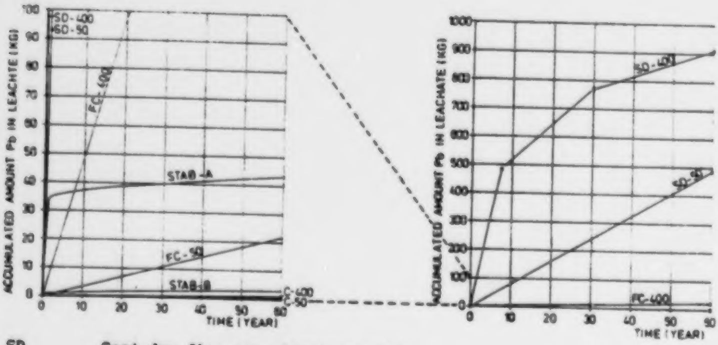
- The average height of the deposit of the non-stabilized residues is 3 metres. The exposed "precipitation area" will therefore vary according to the amount of residue produced.

- The cement-stabilized residue is deposited in accordance with one of the following alternatives:

  A - deposition of cubes 1m x 1m x 1m
  B - deposition of monolithic blocks with an exposed area of 3 500 m^2

- The net infiltration through the unstabilized residues has been set at 50 mm/year (applicable Swedish recommendations) and 400 mm/year (normal net infiltration in soil)

- The time horizon is limited to 60 years, i.e. about 2 generations.



SD	Semi-dry flue gas cleaning residue
C	Condensing flue gas cleaning residue
FC	Fluidized limestone bed (RDF)
STAB-A	Cement-stabilized semi-dry flue gas cleaning residue, "cube deposition"
STAB-B	Cement-stabilized semi-dry flue gas cleaning residue "monolithic"

Figure 6. Calculated leaching of lead from flue gas cleaning residues from the semi-dry and the condensing processes and the fluidized limestone bed, as well as the semi-dry residue with cement stabilization. Deposition without sealing (400 mm/year) and conforming to applicable Swedish recommendations (50 mm/year).

Leaching varies widely over time and well motivates different design of the deposition method for the individual residues.

CONCLUSIONS

The results from the chemical and physical characterisations of the residues from the four different flue gas cleaning processes and the environmental scenario modelling may be summarised as follows:

- The chemical/technical flue gas cleaning processes are different.
- The variation in solid phase content of elements is small, at most a factor 5.

- Leachate from all residues have a high pH.

- Leaching of chloride is 2-30 times greater from SD and SDC at L/S=1 than from C and FC. Chloride is easily leached, 100% at L/S=16, from all end products.

- Metal concentrations are generally higher in leachate from SD and SDC than from C and FC, especially for lead and zink.

- The leachable fraction of the solid phase content is higher for SD and SDC in comparison with C and FC. With a solid phase content of 25% Ca and a leachable fraction of 50% the loss of material is great enough to cause subsidiences.

- Good compaction results and low permeabilities are achieved with the semi-dry residue if compaction is performed immediately after moistening. Since leaching from the residue is dependent on permeability, this measure is important. In loose compaction, which normally occurs at the deposition sites, the permeability is high (10^{-4} m/s).

- Tests with stabilization of the semi-dry residue show that it is possible to obtain a high reduction in leaching of lead. Effective stabilizers include water, cement, bentonite, sodium metasilicate and vermiculite.

- In mixing fly ash and sludge from condensing flue gas cleaning, good chemical stabilization of Cd, Pb and Zn is obtained. Leaching of metals is generally very low in the case of C.

- Production of flue gas cleaning residues per ton incinerated refuse varies between 11-70 kg/ton, according to the cleaning process.

- Pollution by lead from the deposition of an annual production of flue gas cleaning residues from a plant of the size of 200 000 ton incinerated waste per year varies widely between the various residues. This variation motivates different deposition techniques to obtain a comparable level of pollution.

- The difference in leaching when sealing to the net infiltration 50 mm/year (applicable requirement on deposition) compared with 400 mm/year (natural infiltration in soil) is small in relation to the differences in the total leaching of lead from the differ- ent flue gas cleaning residues to the environment.

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DEVELOPMENT OF A LABORATORY TEST METHOD FOR ESTIMATING LEACHATE QUALITY FROM MUNICIPAL INCINERATOR ASH MONOFILLS

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ABSTRACT

The U.S. Environmental Protection Agency's (EPA's) existing leachate quality estimation test procedures, Extraction Procedure (EP) and Toxicity Characteristic Leaching Procedure (TCLP) were developed as tools for waste classification and are designed to predict the average long term quality of leachate resulting from co-disposal of small amounts of industrial waste in a sanitary landfill. The tests were developed assuming that the disposal site contains 95 percent municipal waste and 5 percent industrial waste. This is intended to model the "worst-case" mismanagement scenario of disposing a hazardous waste in a sanitary waste landfill.

The assumptions on which these tests are based may not apply to all waste disposal scenarios. For example, a waste that is produced in very large quantities is often disposed in facilities dedicated solely to that particular waste. These facilities are called monofills. In a true monofill the waste does not contact leachate characteristic of that generated by sanitary refuse decomposition, but rather, self-generated leachate (i.e., leachate formed by contact of ambient water with the waste). Therefore, using either the EP or TCLP could result in an unrealistic estimation of leachate quality from a monofill. To address leachate predictions from other disposal scenarios, the EPA/Office of Solid Waste (OSW) has begun a broad-scope, long-term method development program.

The purpose of this paper is to present results from laboratory tests that evaluate important parameters used to predict the leaching of municipal incinerator ash. The first parameter that was evaluated was leaching fluid. The approach used was to perform batch extractions on ash using different extraction fluids. The fluids that were selected were synthetic acid rain, deionized water, and TCLP extraction fluid No. 2, acetic acid.

Additional laboratory work included titrations and multiple extractions of fly ash with synthetic acid rain to determine the amount of acid required to exhaust the ash acid-neutralizing capacity. The titrations were conducted using unaltered fly ash and ash pulverized to less than 80 microns.

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INTRODUCTION

To predict the composition of leachates from municipal waste incinerator (MWI) ash disposal sites and other monofills where the EP and TCLP may be inappropriate predictive tests, the EPA/OSW has begun a broad-scope, long-term program. The initial goals of this program are to develop methods or method modules ranging from those designed for site-specific conditions to those for the more general cases of the disposal of a particular waste type.

The development of these test methods and modules will involve identification of the appropriate waste sample preparation procedure, leaching fluid, leaching fluid contacting procedure, contact regime (continuous vs. wet/dry, single vs. multiple extraction), and leachate collection and analysis methods.

The first such scenario for which a test method is being developed is the monofilled disposal of municipal waste incinerator ash. Under a work assignment for OSW, RTI initiated this effort. The first activity was to assess the currently available information on leaching tests, monofills, and municipal incinerator ash.

Industry contacts were made to determine the state-of-the-art in MWI ash disposal and leaching. Contacts included incinerator operators, EPA representatives, and researchers in the fields of leaching, incineration, and municipal solid waste. Also contacted were a limited number of MWI operators. Discussed during these contacts were incinerator feed materials, ash disposal practices, disposal site characteristics, and the operator's willingness to participate in EPA's study by providing ash or leachate samples. The results from these contacts were presented in a report to EPA.[1]

After review of the literature and information obtained through industry contacts, it was determined that one of the variables having the greatest effect on the leaching of municipal incinerator ash was the leaching fluid. Because this variable has such a large effect on the results of leaching tests, it was selected to be studied first. Several different fluids have been used by researchers. Three of these fluids (synthetic acid rain, deionized water, and TCLP extraction fluid No. 2) were chosen for use in preliminary batch extractions of municipal waste incinerator fly ash.

LABORATORY TESTING

PHASE I - EVALUATION OF LEACHING FLUIDS

Phase I - Procedure

Using the literature review and expectations regarding the behavior of fly ash when contacted with various leaching fluids, several different fluids were investigated in laboratory tests.

It was predicted that the fly ash buffering capacity will control the release of metals into solution. To confirm this, the effect of the leaching fluid must be isolated from other variables. The EP, TCLP, and Monofilled Waste Extraction Procedure (MWEP) involve similar, but not identical conditions. To limit the effect of these method variations on metal release into solution, the TCLP was selected as the test method to confirm this hypothesis. Extractions of municipal incinerator ash were performed with different extraction fluids along with the equipment and conditions of the TCLP. The fluids selected were synthetic acid rain, deionized water, and a TCLP extraction fluid (determined by testing the waste as per the TCLP).

Synthetic acid rain was chosen because it is likely to be most representative of the liquid that would come in contact with ash in a monofill by way of infiltration. Because ash is highly alkaline, it may be capable of neutralizing the synthetic acid rain. Therefore, deionized water was also selected as an extraction fluid to evaluate the difference in extraction potential between this medium and synthetic acid rain. Finally, the TCLP extraction fluid was used to determine if because of the complexing of elements by the acetate ion, the resulting concentration of metals in this extract would be much higher than in either of the other fluids, irrespective of the pH in the final extract.

The synthetic acid rain (deionized water adjusted to pH 3.0 ± 0.2 using 60% H_2SO_4/40% HNO_3 w/w) was prepared to simulate a worst-case precipitation mixture falling on a facility and infiltrating into the monofill. The choice of the TCLP extraction fluid as determined by the crude alkalinity test in the TCLP was fluid No. 2, acetic acid, ~pH 3. The ash pH from this test was barely greater than 5.2 as called for in the method. Because the test involves mixing the ash with water in a beaker for a short period of time, this low pH most likely resulted from the release of the adsorbed acids from the ash surface. Over a longer reaction period, the buffer capacity of the waste would likely neutralize the acid rain medium.

Should the results show no difference between the leachate produced by the synthetic acid rain and deionized water, the deionized water could be used as the extracting medium for these wastes in future studies. This would also make unnecessary the use of site-specific extraction media, as the acid rain employed in the comparison study is more acidic than any acid rain reported in the United States.

Samples of fly ash were obtained from an EPA contractor who had collected the ash from a municipal solid waste incinerator at different times on the same day.[2] These samples were contained in two glass bottles. Before any tests were conducted, the contents of both bottles were combined and tumbled to make one homogeneous sample. The combined ash was then weighed so that the maximum amount of sample that was available to be used in each extraction could be determined. Aliquots of the ash were then taken for initial metals determination and for the extractions. The liquid-to-solid ratio of 20:1 by weight, the extraction period of 18 h, and the hardware used to carry out the extraction were as specified in the TCLP. The only deviations from the procedure were the use of the three different fluids and the use of only 60 g of ash per extraction because of insufficient quantities of the material. Three replicate extractions and one blank were performed for each of the extraction fluids. Four bottles were extracted simultaneously, three containing ash and one containing the extraction medium blank. A total of three extraction runs were carried out to perform all of the extractions. The pH of the filtered extracts was determined and the extracts were then acidified to preserve them for elemental analysis. Elemental analyses were performed on the extracts and on the bulk ash prior to extraction using inductively coupled argon plasma atomic emission spectroscopy (ICP-AES) and atomic absorption spectrophotometry. The bulk ash was digested according to SW-846 Method 3050.

Phase I - Results

The analyses of the bulk unextracted ash (Table 1) showed a composition typical of that often reported for municipal waste incinerator ash. The nitric acid digestion of Method 3050 is not intended to completely dissolve the sample. Those elements that are part of a silicate matrix, such as aluminum and silicon (concentration not determined), are probably present at much higher levels than the ash concentrations reported. Zinc, calcium and iron were present at percentage levels. The high calcium level again suggests the presence of a highly alkaline matrix.

Most notable are the levels for the elements most often considered toxic when in soluble form: barium (~1,000 µg/g), cadmium (~400 µg/g), chromium (~120 µg/g), lead (~5,000 µg/g), nickel (~90 µg/g) and arsenic (~70 µg/g). If only a small percentage of these elements were to be leached out of the ash into ground water, toxic levels might be reached.

Also notable is the excellent repeatability of the analyses of the bulk ash. The duplicate determinations generally agreed to better than 10 percent for these elements. This suggests that this particular ash sample was homogeneous with respect to these elements. This is not always the case with these materials.

The pH's of the ash extracts (Table 2) show the effect of the ash alkalinity on the original extracting fluid. The final pH of the deionized water (original pH around 7) and acid rain (original pH around 3) both increased to approximately 11. The ash alkalinity more than neutralized the acid rain. The pH of the TCLP fluid No. 2 increased from approximately 3 to 6.5. This much lower final pH is due to the acid buffering of the acetate

TABLE 1. ELEMENTAL COMPOSITION OF BULK ASH
(µg/g or %)

Element	Replicate 1	Replicate 2	Mean ± s.d.
Calcium (Ca) (%)	7.3	6.5	6.9 ± 0.6
Aluminum (Al) (%)	4.4	4.4	4.4 ± 0
Zinc (Zn) (%)	2.1	1.9	2.0 ± 0.1
Iron (Fe) (µg/g)	9600	8670	9,170 ± 707
Lead (Pb) (µg/g)	5700	5480	5,590 ± 156
Barium (Ba) (µg/g)	1080	975	1,028 ± 74
Copper (Cu) (µg/g)	821	782	802 ± 28
Cadmium (Cd) (µg/g)	434	417	426 ± 12
Chromium (Cr) (µg/g)	118	123	121 ± 4
Vanadium (V) (µg/g)	88	87	87 ± 1
Nickel (Ni) (µg/g)	82	90	86 ± 5
Arsenic (As) (µg/g)	74	68	71 ± 4
Antimony (Sb) (µg/g)	141	28	84.5 ± 56.5
Selenium (Se) (µg/g)	16	16	16 ± 0
Beryllium (Be) (µg/g)	0.4	0.4	0.4 ± 0

TABLE 2. pH OF EXTRACTING FLUIDS BEFORE AND AFTER 18-h EXTRACTION WITH FLY ASH

Fluid	Before	After[a] 1	After[a] 2	After[a] 3
Deionized water	7.3	11.01	10.96	11.01
Acid rain	3.04	10.95	10.95	10.97
Acetic acid	2.96	6.52	6.57	6.54

[a] 3 replicates.

ions in this system. The agreement in the pH among the replicates for each fluid also supports the bulk waste elemental analysis data in suggesting that the waste was homogeneous.

The results from the extractions of ash using each fluid are given in Tables 3, 4, and 5 and are summarized for all fluids in Table 6. Detection limits for the elements in the extracts are given in Table 7.

In general the results show that very little of the elements were extracted by the deionized water and acid rain fluids. In several cases the level in the extract was nearly the same as in the blank, suggesting no measurable extraction. (The blank results reported in each case were based on analyses of extracting fluids prepared and carried through the procedure without fly ash.) The final extract concentrations for these two fluids are almost identical, as are the results for individual replicates. These data show that the ash was homogeneous with respect to extractable elements as well as total elements.

The acetic acid fluid dissolved much greater quantities of some, but not all, elements than did either the deionized water or acid rain. Concentrations were nearly the same in all extracts for aluminum, antimony, barium, beryllium, chromium, iron, and selenium. Levels were much higher in the acetic acid extract for cadmium, copper, lead, nickel, vanadium, and zinc. The high levels are probably due to a combination of the complexing of these elements with acetate ion plus the added effect of a lower final pH due to the acetate buffering. For example, based on the compositional analysis of the ash, almost 100 percent of the cadmium was extracted by the acetic acid.

PHASE II - MULTIPLE EXTRACTIONS

Phase II - Procedure

The initial laboratory extraction of municipal incinerator fly ash with a worst-case synthetic acid rain during Phase I produced an extract almost identical to the deionized water extract. This type of result was hypothesized before the extractions were conducted. One of the primary objectives of the Phase II experiments was to determine the number of extractions with this acid rain mixture required to exhaust the acid neutralizing capacity of the fly ash. When this point was reached it was theorized that the pH of the extract would decrease and therefore solubilize metals present in the ash. The amount of extract required to reach this point could then be related to the breakthrough time for ash disposed in a landfill and the liquid-to-solid ratio necessary to simulate similar conditions in the laboratory.

Two of the residues from the synthetic acid rain extractions conducted in Phase I were selected for the multiple extraction experiments. These residues were dried and weighed and the extractant volume required to produce a liquid-to-solid ratio of 20:1 was determined. The dried residue was placed in the extraction vessels and extracted for 18 h with synthetic acid rain of approximately pH 3. The extract was then filtered and its pH

TABLE 3. RESULTS OF EXTRACTIONS OF FLY ASH WITH DEIONIZED WATER, mg/L

Element	Blank	Replicate 1	Replicate 2	Replicate 3
Al	0.124	0.822	0.917	0.677
Sb	0.056	0.091	0.032	0.117
Ba	0.005	0.263	0.266	0.239
Be	ND	ND	ND	ND
Cd	0.003	0.002	0.002	ND
Cr	ND	0.079	0.086	0.096
Cu	0.036	0.035	0.032	0.024
Fe	0.045	0.034	0.018	0.018
Pb	ND	ND	ND	ND
Ni	ND	ND	ND	ND
V	ND	0.003	0.004	0.004
Zn	0.037	0.075	0.069	0.055
Ca	ND	628	622	689
As	ND	ND	ND	ND
Se	ND	0.018	0.017	0.018

ND = not detected; see Table 7.

TABLE 4. RESULTS OF EXTRACTIONS OF FLY ASH WITH SYNTHETIC ACID RAIN, mg/L

Element	Blank	Replicate 1	Replicate 2	Replicate 3
Al	0.252	0.966	0.678	0.463
Sb	ND	0.069	ND	ND
Ba	0.033	0.245	0.299	0.275
Be	ND	ND	ND	ND
Cd	ND	ND	0.002	ND
Cr	ND	0.092	0.075	0.071
Cu	0.036	0.040	0.025	0.017
Fe	0.031	0.028	0.024	ND
Pb	ND	0.011	ND	0.004
Ni	ND	ND	ND	ND
V	ND	0.002	0.004	0.002
Zn	0.112	0.123	0.076	0.069
Ca	0.11[illegible]	670	693	685
As	ND	ND	ND	ND
Se	ND	0.013	0.012	0.014

ND = not detected.

TABLE 5. RESULTS OF EXTRACTIONS OF FLY ASH WITH TCLP FLUID NO. 2, ACETIC ACID

Element	Blank	Replicate 1	 2	 3
Al	ND	0.209	0.755	0.867
Sb	0.040	0.224	0.052	0.186
Ba	0.041	0.379	0.385	0.378
Be	ND	ND	ND	ND
Cd	ND	20.0	18.2	18.2
Cr	ND	ND	0.026	ND
Cu	0.005	0.300	0.296	0.326
Fe	ND	ND	0.029	0.017
Pb	0.030	12.0	10.6	10.7
Ni	ND	0.237	0.290	0.294
V	ND	0.474	0.456	0.462
Zn	0.076	563	352	357
Ca	ND	1470	1310	1300
As	ND	ND	ND	ND
Se	ND	0.013	0.012	0.014

ND = not detected.

TABLE 6. COMPARISON OF MEAN EXTRACT CONCENTRATION[a] OF ELEMENTS IN FLUIDS TESTED

Element	Bulk ash	Deionized water (mg/L)	Acid rain (mg/L)	Acetic acid (mg/L)
Al	4.4%	0.805	0.702	0.610
Sb	28-141 µg/g	0.080	0.069	0.154
Ba	1028 µg/g	0.256	0.273	0.381
Be	0.4 µg/g	ND	ND	ND
Cd	426 µg/g	0.002	0.002	18.8
Cr	121 µg/g	0.087	0.079	0.026
Cu	802 µg/g	0.030	0.027	0.307
Fe	9170 µg/g	0.023	0.026	0.023
Pb	5590 µg/g	ND	0.008	11.1
Ni	86 µg/g	ND	ND	0.274
V	87 µg/g	0.004	0.003	0.464
Zn	2.0%	0.066	0.089	424
Ca	6.9%	646	683	1360
Se	16 µg/g	0.018	0.013	0.013
As	71 µg/g	ND	ND	ND

[a] Not blank corrected.

TABLE 7. DETECTION LIMITS FOR ELEMENTS IN EXTRACTS, mg/L

Element	Concentration
Al	0.012
Sb	0.075
Ba	0.004
Be	0.0005
Cd	0.002
Cr	0.008
Cu	0.005
Fe	0.010
Pb	0.025
Ni	0.020
V	0.002
Zn	0.024
Ca	0.040
As	0.004
Se	0.004

determined. Approximately 50 mL of the filtered extract was acidified and retained for future analysis. The filter cake was dried to a constant weight, its final weight determined, and the volume of fresh extractant (at 20:1 mass ratio) needed for the next extraction calculated. This procedure was repeated a total of nine times (including the Phase I extraction).

Phase II - Results

Table 8 presents the extract pH, extractant pH, and extraction number for each replicate. As can be seen from the table, the extract pH was approximately 11 for both samples after the first extraction. This pH decreased slightly with each subsequent extraction. After a series of nine extractions the pH had dropped only to approximately 9.4. The original intent of this experiment was to conduct extractions until a pH of less than 7 was obtained. However, due to time constraints, the work was discontinued after nine extractions. At this point the total liquid-to-solid ratio was 180:1.

PHASE III - EFFECT OF ASH PARTICLE SIZE

Phase III - Procedure

Researchers have shown that extraction of fly ash of different particle sizes results in extracts with different properties and constituent concentrations. The purpose of this phase of the research was to determine the effect of particle size reduction on the acid neutralizing capacity of fly ash samples.

A sample of fly ash was pulverized at 20,000 rpm to less than 80 microns using a Sybron/Brinkmann micropulverizer with a nominal 0.13 mm screen. Two replicate 1-g samples of the pulverized fly ash were placed in flasks containing 50 mL of the synthetic acid rain mixture. Two replicate 1-g samples of unpulverized ash were placed in flasks with 50 mL of synthetic acid rain. The flasks were then stoppered, placed on a wrist action shaker, and shaken for 24 h. The pH of the unfiltered extract was then determined. The unfiltered extract was then titrated to an acidic pH with the synthetic acid rain mixture. The volume of synthetic acid rain required to reach this point was recorded.

Phase III - Results

The results from the pH determination of the unfiltered extracts after 24 h on the wrist action shaker are presented in Table 9. As seen from the table, the pH of each of the unfiltered extracts was above 10. In both cases the pH of the extract from the pulverized ash sample was slightly lower than that from the as-received samples. Table 10 presents the pH of the unfiltered extracts as a function of the amount of titrant added during the second part of this experiment. These data are presented in terms of the milliliter titrant added per milliliter of the unfiltered extract and the milliliter titrant added per gram of fly ash. The final mixtures from the second fly ash sample were allowed to settle for 24 h. At the end of

TABLE 8. MULTIPLE EXTRACTIONS OF FLY ASH WITH SYNTHETIC ACID RAIN

	Fly ash #1		Fly ash #2	
Extraction number	Extractant pH	Extract pH	Extractant pH	Extract pH
1	3.04	10.95	3.04	10.95
2	3.07	10.5	2.92	10.6
3	2.85	10.1	2.85	10.2
4	3.03	10.0	3.03	10.1
5	3.04	10.1	3.04	10.2
6	3.03	9.9	3.03	10.1
7	3.00	9.75	3.00	10.0
8	3.02	9.52	3.02	9.92
9	3.02	9.46	3.02	9.42

TABLE 9. EXTRACT pH AS A FUNCTION OF ASH PARTICLE SIZE

	Ash #1		Ash #2	
	As received	Pulverized	As received	Pulverized
pH	10.2	10.1	10.5	10.1

TABLE 10. TITRATION OF FLY ASH EXTRACTS

mL titrant / mL unfiltered extract	mL titrant / g fly ash	As-received fly ash #1	As-received fly ash #2	Pulverized fly ash #1	Pulverized fly ash #2
0	0	10.2	10.5	10.1	10.1
2	100	9.3	9.1	8.9	9.0
3	150	9.3			
4	200		8.3	8.1	8.0
5	250	9.2			
6	300		6.1	6.7	7.1
7	350	8.9			
8	400		5.6	6.2	6.4
9	450	8.9			
10	500		5.3	5.9	6.0
11	550	8.2			
12	600		5.0	5.5	5.7
13	650	7.0			
14	700		4.5[a]	5.4	5.5[b]
15	750	6.7			

[a] Same mixture 24 h later: settled pH = 6.29
stirred pH = 6.53

[b] Same mixture 24 h later: settled pH = 6.53
stirred pH = 6.7

this time the pH values of both the settled and stirred solutions were determined. These data are also shown in Table 10.

As can be seen from Table 10, the amount of synthetic acid rain titrant required to lower the pH of the extract to below neutral was approximately the same for three of the samples. The remaining sample (fly ash No. 1, as received) required twice as much titrant to reach the same point. In this experiment there does not appear to be any correlation between the acid neutralizing capacity of the ash and the ash particle size. It is interesting to note that for the samples that were allowed to settle for 24 h, the pH in both cases rose between 1 and 2 pH units, thus indicating that the ash still had more acid neutralizing capacity.

CONCLUSIONS

The results of these limited laboratory studies suggest that the alkalinity of municipal waste incinerator fly ash probably controls and determines the release of elements into leachate until this alkalinity is exhausted. For this reason, distilled water is as effective a leaching fluid as a worst-case synthetic acid rain. Thus, a site-specific leaching fluid is probably not needed for testing such wastes. With some exceptions (most notably chromium), the acetic acid extractant is more aggressive than either the deionized water or the synthetic acid rain. In the case of cadmium, the acetic acid dissolved 88 percent of the metal presented in the ash in a single 20:1 extraction.

The multiple extractions of fly ash using synthetic acid rain and the TCLP test conditions showed that the extract pH decreased slightly with each subsequent extraction. After a total of nine sequential extractions, the pH of the extract had decreased approximately 1.5 pH units to 9.4. This indicates that the acid-neutralizing capacity of the fly ash is very high and slow to be depleted.

An unanswered question from this research is whether the alkalinity can be depleted under landfill conditions so that the release of toxic elements is no longer inhibited. This should be addressed in laboratory tests involving additional multiple extractions as well as in column or lysimeter studies where the effects of variable liquid-to-solid ratios can be evaluated.

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PUBLIC HEALTH RISKS ASSOCIATED WITH DISPOSAL OF MSW INCINERATION ASH

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ABSTRACT

Use of incineration as a technique for the volume reduction of MSW prior to disposal has become widespread in the United States and overseas. Considerable effort has been applied to characterizing the health risks associated with stack emissions from these facilities. However, much less attention has been given to evaluating the ultimate disposal of the ash residue. Such an evaluation is necessary in order to address the cumulative impact of the incineration option.

The evaluation of health risks of ash disposal must include consideration of fugitive emissions to the air, contamination of groundwater, and contamination of surface water. This paper will review the techniques available for characterizing the public health risks associated with disposal of incinerator ash. This will include methods for estimating the release of contaminants to the environment, as well as the environmental fate of these releases. Exposure pathways to man will be examined. Specific examples of the results of risk assessments performed for ash monofills will be presented, with emphasis on those factors that most strongly affect the quantitative risk estimates. The quantitative results of risk assessmens for ash monofills will be put into perspective through comparison to the risks associated with landfilling of raw MSW. Mitigation measures for minimizing human exposure to the toxic components of ash will be identified.

INTRODUCTION

In recent years, a considerable amount of effort has been given to characterization of the potential health impacts associated with the stack emissions resulting from municipal solid waste incineration. However, to date, very little effort has been expended on the development of quantitative health risk estimates for incinerator ash disposal. One of the reasons for this may be the fact that it is more difficult to characterize the risks associated with ash disposal. This is primarily due to the greater difficulty in characterizing the magnitude of pollutant releases from ash disposal sites. It is relatively easy to sample the stack emissions from a municipal solid waste incinerator. However, the emissions from the handling and disposal of ash are much more intermittent in nature and tend to occur over a larger physical area. The issue of contaminated leachate and transport of this leachate to and through groundwater presents numerous technical challenges in the assessment of exposure and risk. Groundwater contamination resulting from incinerator stack emissions has not been considered to be an important issue (1).

This paper will summarize some of the technical issues associated with the preparation of quantitative health risk assessment for the disposal of ash. The discussion will focus on the disposal of ash in monofills, as this is the strategy that is most often proposed for new waste incineration projects. A case study of a hypothetical ash monofill has been constructed in order to illustrate some of the issues associated with human exposure.

ELEMENTS OF RISK ASSESSMENT

The approach and individual assumptions used in preparation of health risk assessments can differ considerably depending on regulatory agency requirements as well as the professional judgment of the authors. There is no standardized, consistent method for risk assessment preparation. However, the basic elements to be included in an evaluation of health risks are fairly well established, and have been defined by EPA within published guidelines (FR 51: 33992). EPA defines risk assessment to have four major steps: hazard identification, dose response assessment, exposure assessment, and risk characterization. However, for the purpose of describing the actual steps taken in preparation of a health risk assessment, it is most informative to define six major tasks. These steps include the characterization of pollutant releases, hazard assessment, dose/response determination, environmental transport and fate analysis, exposure assessment, and risk characterization.

The initial step in an evaluation of health risks is to determine the pollutant emissions and/or effluents from the facility or activity in question. This can be established through direct measurements at similar facilities or through published emission estimates. A hazard assessment is usually prepared for the pollutants that will potentially be released to the environment. This consists of a qualitative disclosure of the potential consequences of exposure to the substances being evaluated. The hazard assessment is performed through a review of the pertinent animal experimental data and human epidemiological studies.

A dose/response determination is performed in order to estimate the level of human exposure required to produce the toxic effects presented in the hazard assessment. Substances that cause cancer are assumed to have no lower threshold of effect. In other words, this implies that any lifetime exposure to a carcinogen produces a finite risk of cancer. At low doses, this probability is assumed to be linear with dose. Low-dose potency slopes published by EPA's Carcinogen Assessment Group (CAG) are generally used in computing quantitative cancer risk estimates. These potency slopes have been developed on the basis of both human and animal data. In evaluating noncancer health impacts, the operating assumption is that an effects threshold does exist. If human exposure to a substance is less than established effect thresholds, then adverse health impacts of a noncarcinogenic nature are unlikely.

In order to evaluate human exposure to the emissions/effluents from a project, it is necessary to model their transport and eventual fate in the environment. Figure 1 is a graphic example of this for an incineration project. It shows that it is necessary to determine all of the mechanisms that will cause pollutant transport into the various environmental media. Of particular concern are the air, water (both groundwater and surface water), soil, and biota (for purposes of foodchain exposure considerations).

Based on the estimate of the concentration of toxics in each environmental compartment, the human intake of these substances is calculated. This occurs through three major routes: inhalation, ingestion, and through the skin following direct contact (dermal absorption). There are several different types of ingestion pathways including food, drinking water, and hand-to-mouth soil ingestion. Exposure routes considered in health risk assessment are shown in Figure 2.

The final step in the risk assessment preparation process is termed risk characterization. Cancer risk is presented as the incremental lifetime increase in the probability of contracting cancer as a result of a given project. Noncancer impacts are addressed by the comparison of background and facility-related exposure to effect thresholds. Often, the ratio of threshold to exposure is presented as a safety factor. An additional part of risk characterization that has been identified by the EPA in their guidelines on risk assessment is a disclosure of the uncertainty involved in the quantitative risk estimates.

RISK ASSESSMENT FOR ASH MANAGEMENT

The procedure for conducting risk assessment for an ash monofill would be identical to that described above. The major difference between a risk assessment for an ash monofill and the stack emissions from a waste incinerator would be in the evaluation of environmental transport and fate of pollutant releases. Figure 3 illustrates the major contaminant release and transport scenarios for an ash monofill. These include the fugitive emissions of contaminant-laden ash to the atmosphere, the contamination of groundwater resources by leachate escaping the ashfill, and the transport of contaminated liquids to local surface waters due to runoff. Figure 3 also serves to illustrate the type of worst-case, health-protective scenario that is often used in the preparation of generic risk estimates. In actuality, siting and engineering considerations would do much to mitigate the apparently disastrous situation illustrated in Figure 3. In the preparation of a site-specific risk assessment, it is advisable to account for those specific factors associated with the surrounding environment as well as the ashfill design and operating plan. This could include considerations such as the location of nearest residences, proximity to groundwater and surface water resources, use of these water resources, and the use of liners and leachate collection systems.

An important consideration in evaluating the risks are the characteristics of the ash. Table 1 shows the average concentration of several contaminants in bottom ash, fly ash, and combined ash. It is evident that these contaminants are considerably more concentrated in the fly ash than in bottom ash. Fly ash also has a much greater proportion of fine particles and is thus, more likely to become airborne and emitted to the atmosphere. The characteristics of combined ash are much more similar to bottom ash than fly ash due to the fact that the total production of ash at a waste incinerator is approximately 90 percent bottom ash and 10 percent fly ash.

Another important consideration is the concentration of metals in the leachate produced in the ash monofill. Table 2 presents the observed range of metal concentrations in actual leachate from ash monofills. As can be seen in this table, the actual leachate quality is frequently much better than that simulated through extraction tests such as the EP Toxicity test. Cadmium and lead, which frequently are the cause of the failure of combined ash samples to meet the EP Toxicity test, have much lower concentration in actual monofill leachate. In many cases, the measured quality of actual monofill leachate has been observed to meet drinking water standards. If possible, it would be preferable to rely on data from operating monofills to establish leachate quality for the purposes of quantitative risk assessment. The extraction tests, such as EP Toxicity, do not simulate conditions within an ash monofill and hence, will not accurately predict the quality of actual leachate (2, 3, 4).

The evaluation of leachate and its potential for contamination of the groundwater can generally be limited to inorganic substances. This is because the solubility of the most toxic organic substances is extremely low. In addition, many toxic organics such as dioxin and PCBs have an affinity for binding to organic carbon in the soil, and thus being unavailable for transport to the groundwater (5).

CASE STUDY FOR AN ASH MONOFILL

In order to illustrate some of the approaches and issues associated with the performance of health risk assessment from an ash monofill, a case study has been constructed. The following basic assumptions have been used.

- Ash from a 1,000 ton/day waste incinerator in New York State.
- 250 ton/day of combined ash.
- An average of 0.5 acre of exposed ash surface (i.e., no intermittent cover).
- Activities at ashfill include dumping, spreading, and compaction.
- Combined ash and leachate contaminant characteristics are equal to the average defined in Tables 1 and 2.
- Physical characteristics of ash (moisture content, silt content) are equal to the average observed at Marion County, Oregon (6).

The case study will address only the impacts occurring at the ash monofill. The impacts of ash handling and transfer at the incinerator facility should generally be addressed as part of the facility design and permitting. Environmental impacts associated with the transport of ash to the monofill should be minimal and sufficiently widespread to not contribute to potential health risk in a significant manner. It will also be assumed that a stormwater collection and retention system can be engineered such that stormwater runoff from exposed ash surfaces to the environment will not occur.

GROUNDWATER CONTAMINATION

A screening analysis can be conducted to determine if groundwater contamination by leachate from the ashfill could produce significant human exposure to contaminants. This can be accomplished through the use of several conservative, simplifying assumptions.

- Loss of liner integrity produces a continuous point source of contamination.
- There are no attenuation losses of contaminant during transport to or through the groundwater.
- There is uniform groundwater flow.
- Concentrations are calculated at the centerline of the leachate plume.
- Steady-state concentrations.

All of the above assumptions would tend to maximize contaminant concentrations in the groundwater.

Steady-state centerline leachate plume concentrations for two-dimensional horizontal flow can be calculated according to the following equation (7).

$$C(x) = \frac{C_o Q}{b \, 2 \pi p} \left(\frac{\pi B}{2 D_x D_y r} \right)^{1/2} \exp \left(\frac{v_x x}{2 D_x} - \frac{r}{B} \right)$$

where:

$C(x)$ = centerline concentration at distance x

C_o = initial leachate contaminant concentration, (mg/l)

Q = volumetric discharge of leachate, (m^3/day)

b = saturated thickness of mixed zone, (m)

p = effective porosity of the aquifer, (dimensionless)

D_x, D_y = dispersion coefficients in the x and y directions, respectively, (m^2/day)

v_x = seepage velocity of the regional flow in the x direction, (m/day)

r = $[\beta \, (x^2)]^{1/2}$, (m)

β = $1 + 2 B R_d k / v_x$, (unitless)

k = first-order decay constant, (day^{-1})

R_d = retardation coefficient for linear equilibrium adsorption, (unitless)

B = $2 D_x / v_x$, (m)

Note that for the special case of no attenuation, $R_d = 0$, $k = 0$, $\beta = 1$, and $r = x$. This changes the equation to the following.

$$C(x) = \frac{C_o Q}{b \, 2 \pi p} \left(\frac{\pi B}{2 D_x D_y x} \right)^{1/2} \exp \left(\frac{v_x x}{2 D_x} - \frac{x}{B} \right)$$

This approach has been developed into a nomograph (8). This nomograph was used to calculate centerline concentration with distance from the leachate entry into the groundwater. The calculation was conducted using arsenic as an example. The highest arsenic concentration observed in actual leachate (0.218 mg/l) was assumed to be present in the leachate from the hypothetical ashfill (2). The aquifer characteristics were selected on the basis of data collected from an observed contamination incident involving electroplating wastes at South Farmingdale, New York (8). The values of these parameters are as follows.

b = 110 feet

p = 0.35

v_x = 1.5 ft/day

D_x = 105 ft^2/day

D_y = 21 ft^2/day

The maximum quantity of leachate generated can be estimated through use of the following assumptions.

- Total ashfill area of 20 acres, of which 0.5 acre is the working area.
- Annual precipitation of 40 inches.
- Leachate generation is 100 percent of rainfall for uncovered ash, 15 percent for covered ash.

The leachate generation in the working (uncovered) portion of the ashfill is:

(1.0)(0.5 acre)(40 in./yr)(1 ft/12 in.)(1,234 m^3/acre-ft) = 2,055 m^3/yr

The total leachate generated is approximately 14,000 m^3/year; or 10,000 gal/day. Based on this result, three different scenarios of leachate loss were developed: 10,000 gal/day, 1,000 gal/day, and 100 gal/day. This assumes that a liner failure leads to loss of 100 percent, 10 percent, and 1 percent of the total leachate, respectively.

The results of the groundwater concentration calculations appear in Figure 4. For each of these scenarios, the centerline arsenic concentration drops below the primary drinking water standard within 100 feet of the leak. For a small leak (100 gal/day), the concentration drops off by almost 3 orders of magnitude in less than 100 feet. These results would indicate that, on the basis of observed concentrations of contaminants in actual ashfill leachate and the effect of dilution in the aquifer, that groundwater contamination does not appear to present the potential for significant human exposure. This conclusion is considerably strengthened by the extremely conservative assumptions used in the calculations (centerline concentrations, no attenuation, etc.).

FUGITIVE EMISSIONS

The other issue area examined in the case study is that of fugitive emissions of ash due to disposal activities. Several assumptions were made in estimating the magnitude of emissions.

- o The ash is transported and dumped in 12 cubic yard loads.
- o There is no daily cover for the ash.
- o EPA emission factors for aggregate handling, unpaved roads, and wind erosion apply to the ash.
- o Ash silt content = 14.6%.
- o Ash moisture content = 12.5%.
- o Average wind speed = 9 mi/hr.
- o There are 150 days/year of precipitation $\geq$ 0.01 inch.

The first two assumptions are based on operating experience at the Indianapolis ash monofill. Physical characteristics of the ash are average values from the Marion County, Oregon facility (6). Meteorological conditions are typical values for New York State.

The three activities at the ash monofill leading to emissions include the unloading of ash, spreading and compaction, and wind erosion from exposed ash surfaces. Batch drop emissions from unloading trucks can be expressed by the following equation (9).

$$E = \frac{k\,(0.0018)\,(s/5)\,(u/5)\,(H/5)}{(m/2)^{2}\,(y/6)^{0.33}} \quad \text{(lb/ton)}$$

where:

k = particle size multiplier (dimensionless)

s = percent silt content

u = wind speed (mph)

H = material drop height (ft)

y = volume of load (yd^3)

m = percent of moisture

It can be seen from this equation that emissions are maximized by high winds, high drop heights, high silt content, and low ash moisture content. Moisture content is the most important parameter, as it is raised to the second power in the denominator.

Emissions from spreading and compacting activities are estimated as follows (9).

$$E = k(5.9)\,(s/12)\,(vs/30)\,(w/3)^{0.7}\,(nw/4)^{0.5}\,(365\text{-}p/365)(\text{lb/VMT})$$

where:

k = particle size multiplier (dimensionless)

s = percent silt content

vs = vehicle speed (mph)

w = mean vehicle weight (tons)

nw = number of wheels

p = number of days per year with more than 0.01 inch of rain

The magnitude of these emissions depends upon the speed and type of equipment being used as well as the silt content of the ash. It is evident that a high moisture content will reduce emissions. Occasional watering of exposed ash surfaces could be an effective mitigation measure. In calculating these emissions, the operating experience at Indianapolis was used to define the equipment used. These assumptions are as follows.

Vehicle weight = 19 tons

Vehicle speed = 2 mi/hr

Fraction of time on exposed ash surfaces = 65% of an 8-hour day

Number of wheels = 2 (i.e., tracked vehicle)

The final source of emissions is due to wind erosion. Wind erosion emissions are intermittent in nature, occurring during high wind speed events. Emissions can be estimated by the following equation (9).

$$E = 1.7\ (s/1.5)\ [(365-p)/235]\ (f/15)(\text{lb/acre-day})$$

where:

s = percent silt content

p = number of days per year with more than 0.01 inch of rain

f = frequency of wind speeds greater than 12 mph

It is evident that the higher frequency of high wind speeds will increase wind erosion emissions. The frequency of winds greater than 12 miles per hour in this equation should represent the winds that occur at the top elevation of the exposed ash. Therefore, ash disposed above grade will have a greater potential for wind erosion.

It is likely that both the equations for wind erosion and the equation for spreading/compaction greatly overestimate the actual emissions. This is because ash tends to be pozzolonic in nature, particularly from facilities employing dry scrubbing systems (10). This results in the deposited and compacted ash forming a hard, nonerodible surface. EPA has estimated that the application of surface binding agents to exposed areas can reduce wind erosion emissions by 99 percent (9). In some ways, the ash may act as its own surface binding agent.

The results of the emission calculations are presented in Table 3. It is apparent that wind erosion and spreading/compacting emissions contribute most to the total emissions occurring at the ashfill. Overall, the total emissions are quite small. The only substance emitted in excess of 1 lb/year is lead, which has an emission rate of 6.0 lb/year.

To put these emissions into perspective, it is useful to make comparisons to the stack emissions from a waste incineration facility. This is presented in Table 4. The comparison is made between a hypothetical ash monofill which disposes of 250 tons/day of ash and a 1,000 ton/day waste incinerator, which would produce approximately 250 tons/day of ash. Emissions from the 1,000 ton/day waste incinerator were based on actual tests at the Pinellas County, Florida facility Unit #3 (11). This is important because a health risk assessment performed on these measured emissions showed that they were too low to be considered significant from a human health standpoint (12). Table 4 shows that the emissions from the ash monofill are small in comparison to the stack emissions, less than 1 percent for most substances. This would indicate that ash disposal does not add in a significant way to the emissions associated with the management of waste through combustion.

Since the ashfill produces an emission source of considerably different characteristics than the stack emissions from a waste incinerator, it is necessary to consider the downwind ambient impacts of these emissions. Using arsenic as an example, the centerline ambient concentration was calculated downwind of the hypothetical ash monofill and a 1,000 ton/day waste incinerator. This was done with the use of the ISCST model and assumption of a neutral atmosphere and moderate wind speeds (i.e., D stability, 5 m/sec). The results of these calculations are graphically displayed in Figure 5. Because of the nature of the ash monofill emissions (i.e., at ground level), the maximum impacts are immediately adjacent to the source. Despite the fact that the emissions of arsenic from ash disposal are much smaller than the stack emissions, the maximum concentration impact from the ashfill may equal or exceed that from the stack emissions. However, this would only occur within a few hundred feet of the working portion of the ashfill. At a distance of 1 kilometer, the impact from the ashfill would be more than an order of magnitude less than the peak impacts of stack emissions. These results would tend to suggest that the disposal of ash generated by the incineration of waste does not add a significant incremental risk to that associated with the stack emissions. Health risk assessments performed for the stack emissions of waste-to-energy facilities have produced quantitative estimates of maximum cancer risk that are in the range of 1 to 10 chances per million (12, 13, 14, 15, 16). This level of risk has been considered to be insignificant (17).

CONCLUSIONS

As a result of the consideration examined in this paper as well as the case study, a number of conclusions can be made.

- Given observed leachate quality from ash monofills and dilution effects, groundwater contamination does not appear to present a significant potential for human exposure.
- The primary potential for human exposure appears to be via contaminant emissions to the air.
- Emissions are a function of moisture content, management practices, and physical characteristics of ash after disposal.
- The risks associated with ashfill emissions would appear to be low in comparison to stack emissions from waste incineration.
- Existing EPA emission factors are probably inadequate for application to ashfill activities owing to the unique physical characteristics of ash.
- Realistic evaluation of health risks from ash monofills await the availability of field measurement programs at ashfill sites.

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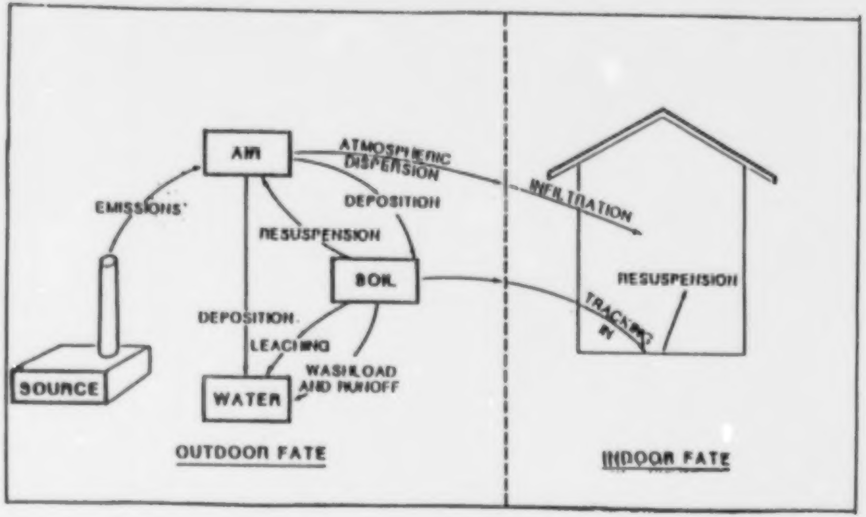
AIR
EMISSIONS
SOURCE
ATMOSPHERIC DISPERSION
DEPOSITION
RESUSPENSION
SOIL
DEPOSITION
LEACHING
WASHLOAD AND RUNOFF
WATER
OUTDOOR FATE
INFILTRATION
RESUSPENSION
TRACKING IN
INDOOR FATE

FIGURE 1. Environmental Fate of Emitted Contaminants.

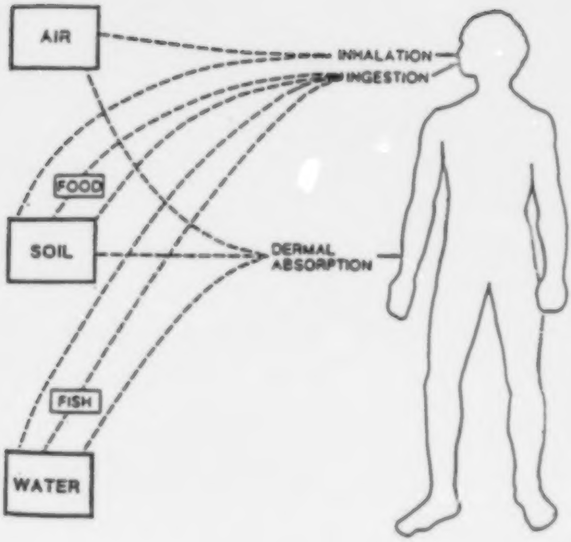
AIR
INHALATION
INGESTION
FOOD
SOIL
DERMAL ABSORPTION
FISH
WATER

FIGURE 2. Routes of Human Exposure to Contaminants.

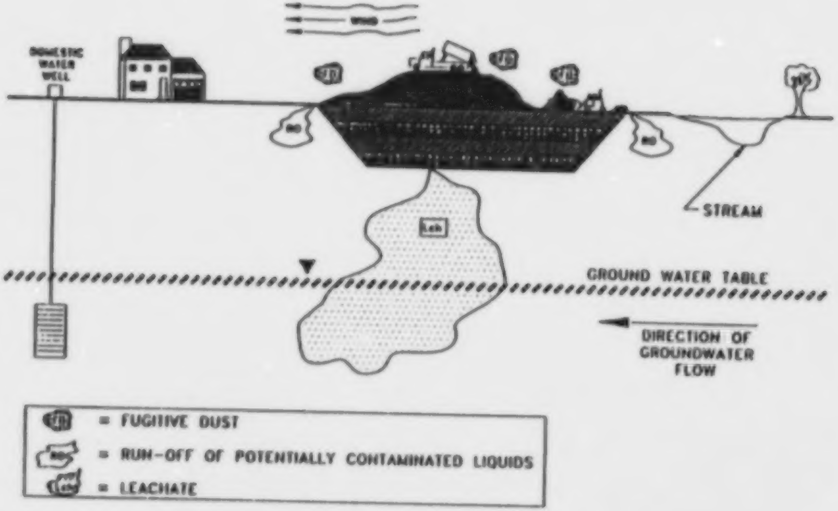
DOMESTIC WATER WELL
WIND
STREAM
GROUND WATER TABLE
DIRECTION OF GROUNDWATER FLOW
= FUGITIVE DUST
= RUN-OFF OF POTENTIALLY CONTAMINATED LIQUIDS
= LEACHATE

FIGURE 3. Examples of Pollutant Release Scenarios at MWC Ash Disposal Sites.

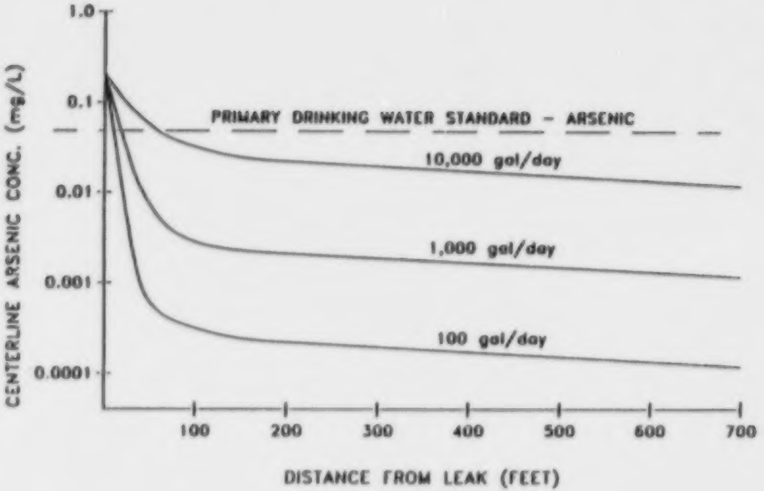

FIGURE 4. Dilution of Leachate in Groundwater.

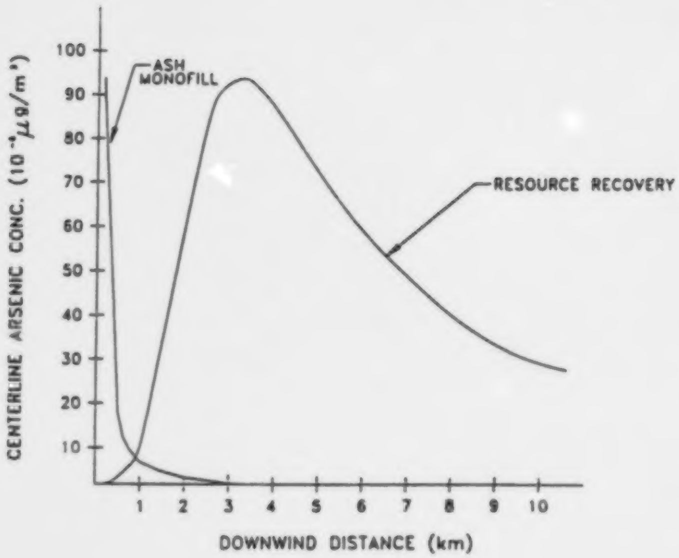

FIGURE 5. Comparison of Downwind Ambient Arsenic Concentrations.

TABLE 1. Contaminant Concentrations in Bottom Ash, Fly Ash, and Combined Ash.

AVERAGE CONCENTRATION OF CONTAMINANTS IN ASH
(mg/kg)

	BOTTOM ASH	FLY ASH	COMBINED
Arsenic	9.2	55.7	7.8
Cadmium	10.7	235	20.6
Chromium	61.2	76.2	46.4
Nickel	41.4	19,104	78.1
Lead	2,800	5,642	2336
PCDD/PCDF	0.021 ug/kg	5.28 ug/kg	0.21 ug/kg

Source: NUS, 1987.

TABLE 2. Ashfill Leachate Quality.

SUBSTANCE	EP TOXICITY RESULTS(mg/l)	ACTUAL MONOFILL LEACHATE(mg/l)	PRIMARY DRINKING WATER STANDARD(mg/l)
Arsenic	0.005 - 0.1	0.005 - 0.218	0.050
Cadmium	0.01 - 3.94	ND - 0.0444	0.010
Chromium	0.0059 - 0.46	0.006 - 1.53	0.050
Nickel	0.241 - 2.43	ND - 0.412	-
Lead	0.02 - 34	0.012 - 2.92	0.050

Note: Results of EP Toxicity tests on combined bottom and fly ash.

Source: NUS (1987)

TABLE 3. Contaminant Emissions from Various Ash Disposal Activities.

ANNUAL AVERAGE EMISSIONS
(lb/year)

SUBSTANCE	BATCH DROP	WIND EROSION	SPREADING/ COMPACTING	EMISSIONS
Arsenic	5.46×10^{3}	1.4×10^{3}	5.4×10^{3}	1.9×10^{2}
Cadmium	1.45×10^{4}	3.8×10^{3}	1.5×10^{3}	5.3×10^{3}
Chromium	3.26×10^{4}	8.1×10^{3}	3.1×10^{3}	1.1×10^{1}
Nickel	5.49×10^{4}	0.144	5.5×10^{3}	0.20
Lead	1.67×10^{3}	4.36	1.67	6.0
PCDD/PCDF	1.47×10^{6}	3.9×10^{6}	1.50×10^{4}	5.4×10^{6}

TABLE 4. Comparison of Stack Emissions and Ash Disposal Emissions.

SUBSTANCE	EMISSIONS(lb/year) ASH MONOFILL	 WASTE-TO-ENERGY[1]	RATIO OF MONOFILL TO WTE FACILITY
ARSENIC	1.9×10^{-2}	10.8	0.0018
CADMIUM	5.3×10^{-2}	24.5	0.0022
CHROMIUM	1.1×10^{-1}	14.1	0.0078
NICKEL	0.20	8.8	0.023
LEAD	6.0	494	0.012
PCDD/PCDF	5.4×10^{-4}	0.27	0.002

[1] BASED ON DATA FROM PINELLAS COUNTY UNIT #3

A JOINT PROBABILITY MODEL FOR PREDICTING CROSS-SECTIONAL RISKS FOR MSW INCINERATOR EMISSIONS OF CONCERN

by

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Paul C. Siebert, Denise R. Alston, and James F. Walsh

Abstract

The site-by-site estimate of the maximum inhalation exposure to any single pollutant emitted from an MSW incinerator requires knowledge of only two factors. These are the pollutant-specific emission rate and the maximum dilution factor (χ/Q) for the appropriate averaging time. There is sufficient dioxin/furan emission data to statistically define the expected frequency distribution of plant-by-plant emissions from future facilities. The same is true for the expected χ/Qs based on the modeling results contained in a significant subset of all plants that will be in operation within the next decade. These two distributions can be combined to produce a joint frequency distribution that can be used to predict the level of inhalation risk and its associated probability for any future facility. The total all-pathway dioxin/furan cancer risk can also be extrapolated from the inhalation model based on the results of several site-specific multipathway risk assessments that we have conducted.

The joint probability model can be applied to any pollutant of concern if sufficient emissions data are available to describe a probability distribution. The model results for dioxins/furans and carcinogenic trace metals show that the probability of cancer risk above levels of any significance is extremely low. This conclusion also applies to lead relative to the ambient lead standard.

Introduction

The performance of health risk assessments has become a routine part of resource recovery project development throughout the nation. In many cases such risk assessments have become a regulatory requirement as part of the required solid waste management permit or as part of the air toxics provisions of new state regulatory programs. Some states now have ambient standards for many of the pollutants emitted by waste-to-energy incinerators.

The Congress, because of political pressure, has been considering the setting of specific pollutant-by-pollutant emissions limitations because of a perceived concern that such limitations are needed to be protective of public health as this technology gains more widespread use. The EPA has projected(1) that the current installed incineration capacity of about 50 thousand tons per day will be tripled to 150 thousand tons per day over the next decade.

In response to the Congressional concerns, the EPA has performed cross-sectional impact analyses(2). The EPA has estimated the carcinogenic risk and other exposure risks associated with the present array of incineration technology. They developed an emissions data base(3) and conducted site-by-site modeling using a generalized dispersion modeling approach. Our review of EPA's risk estimates(4) suggest that their results did not reflect the most likely risks nor give the reviewer a sense of the likelihood of the upper bound risk estimates. A case in point involves the emissions factors used to represent the current installed capacity of mass burn heat recovery facilities. Figure 1 shows the distribution of available plant-by-plant emissions data as well as the average value chosen by EPA to represent the entire

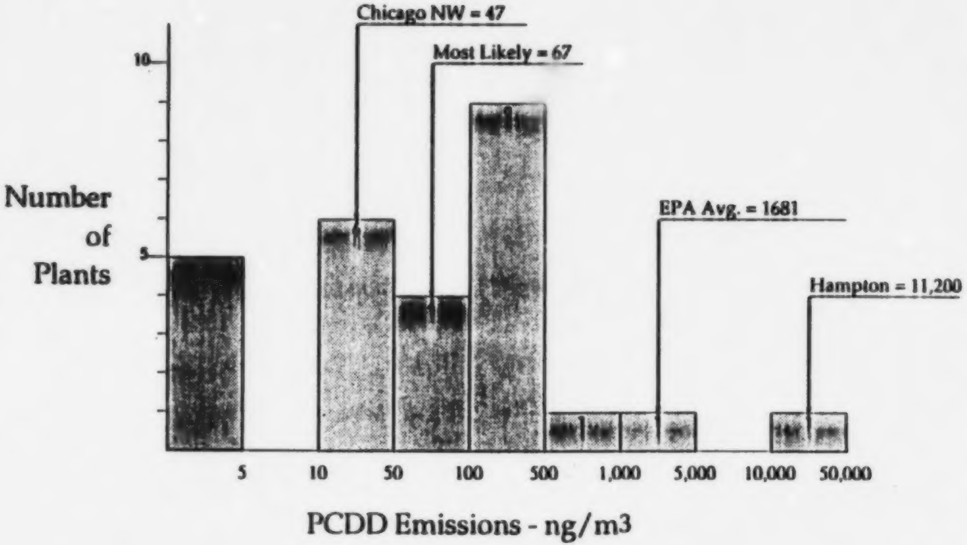


FIGURE 1 Distribution of PCDD Emissions Data for 27 incinerators with heat recovery – worldwide data, all technologies

cross section of existing facilities. The Chicago NW data point is also shown because it was commonly used in earlier risk assessments when little other data was available. We believe that the inclusion of the Hampton data point is innapropriate because of the atypical test conditions as well as the recognized poor combustion design of the incinerator. Further, the simple averaging of the data is inappropriate in light of the demonstrated log normality of the data.(5) A comparison of EPA's risk estimates and those which WESTON derived after making several corrections are shown in Table 1. It can be seen that the EPA risk estimates are significantly higher than those derived by the authors from the same data, but using more appropriate statistical analyses. This same problem was found to be associated with the other technologies and their respective emissions factors as well.

The authors have developed a cross-sectional model(6) that incorporates two data bases, our worldwide emissions data base and a modeling plant capacity/dilution factor data base. Both of these data bases have fixed statistical properties that can be used to predict the exposure risk probabilities for future facilities. The cross-product of the plant capacity/dilution factor distribution and the observed pollutant-by-pollutant emissions distribution yields a joint probability distribution of all possible combinations. The resulting joint probability distribution yields the inhalation risk estimates only. However, there is an expanding multipathway risk assessment data base which will allow for the estimation of total pathway exposures.

Plant Capacity/Dilution Factor Data Base

The authors have collated a series of modeling results from permit applications for both proposed and constructed waste-to-

TABLE 1
Comparison of Risk Estimates
EPA's Report to Congress
versus
WESTON's Corrected Data

Classification	Maximum Individual Risk (chances/million)		Annual National Incidence (cases/year)	
	EPA	WESTON	EPA	WESTON
MSW Non Heat Recovery	5,000	150.	1.3-30.0	1.8
MSW w/Heat Recovery	1,900	3.4	0.2-4.0	.007
RDF	600	7.2	0.1-3.0	.028
Modular w/Heat Recovery	110	6.4	.0008-.01	.001

energy facilities. The cross product of plant capacity times the annual dilution factor yields a frequency distribution representing the probability of all future installation regardless of plant size or meteorological setting. Obviously the larger the data base, the smaller the confidence interval around the data. Table 2 shows the modeling data used in our analysis of the estimated risks associated with future full-size mass burn facilities. Many of these facilities were also modeled by EPA in their report to Congress. As stated earlier, EPA used a more generalized modeling approach which is part of their Human Exposure Model (HEM) algorithm, while WESTON's data is based on site-specific modeling. EPA did not generate a distribution but calculated a risk or exposure value for each plant and then reported the highest values in their summary tables.

Figure 2 shows the comparison of our modeling distribution to the one that we derived using EPA's data base. It can be seen that EPA's result would yield higher risk results by about a factor of 3 at the geometric mean or most likely case. EPA also developed similar capacity and dilution factor data for the other technology classifications, including modular facilities. One might conclude that the individual exposure risks associated with modular facilities would automatically be less than that associated with the larger facilities. This may not be the case because modular facilities are often sited in complex terrain situations and do not employ GEP stacks. Table 3 shows the resulting distributions we developed from the WESTON and EPA data bases. It would appear that modular facilities pose less risk than larger facilities assuming that the emission factors are the same. This may not be the case however.

Table 2
The Annual Dilution Factors and Plant Capacities for Massburn and Refused Derived Fuel Plants Based on WESTON's Database

Plant	Plant Capacity (tons/day)	Annual Dilution Factor (X/Q) ((ug/m^3)/(gm/sec))
Mid Connecticut, CT	2,000	0.0490
Philadelphia, PA	2,250	0.0440
Delaware Co., PA*	2,688	0.0330
Montgomery Co., PA	1,200	0.0350
Montgomery Co., MD	1,800	0.0320
Pennsauken, NJ	500	0.0310
Camden, NJ	1,400	0.0120
Gloucester Co., NJ	575	0.0680
Johnston, RI	750	0.0530
Detroit, MI	3,000	0.0490
Portland, ME	400	0.2300
Los Angeles, CA	1,600	0.1300
Ocean City, NJ	400	0.0880
Tulsa Co., OK	375	0.1400
Broome Co., NY	600	0.2100
Montgomery Co., MD (Dickerson)	2,250	0.0262

FIGURE 2
**Probability of Product of Capacity
times Dilution Factor
for MassBurn and RDF Facilities**

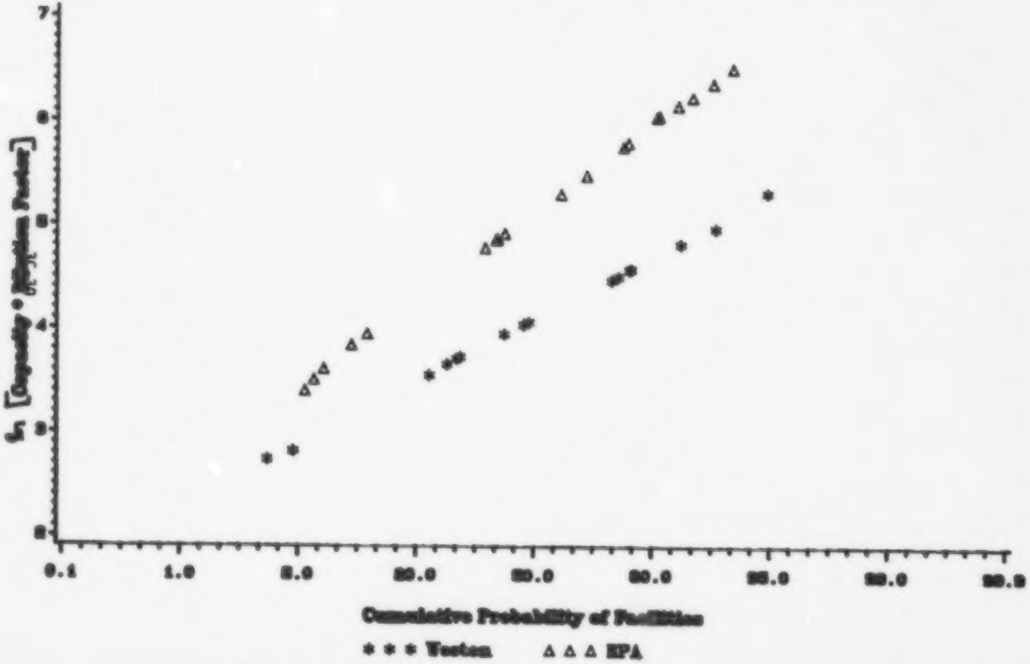


TABLE 3

COMPARISONS OF MODELING DATA BASE DISTRIBUTIONS (TPD x X/Q)

Incinerator Technology	Data Source	Number of Facilities	TPD x X/Q (TPD · ug/m^3 · gm/sec)	
			GM	Upper 95%
Mass burn/RDF Heat Recovery	EPA	19	164	953
	WESTON	16	58	202
Modular	EPA	50	15	51
	WESTON	4	106 (avg)	172 (max)

Emissions Data Base

The emissions data base used in this analysis is taken from Roy F. Weston's worldwide data system. The availab`e emissions data for the dioxins/furans, arsenic, cadmium, chromium, nickel, beryllium, and lead were used in this analysis. The data for dioxins/furans that is representative of the predominant technology being installed today, that includes good combustion design, some form of wet/dry scrubbing, and high efficiency particulate control, is quite adequate to describe the expected probability distribution of the emissions from future facilities. On the other hand, the emissions data base for the remaining pollutants is less reliable. In these cases, the mean or geometric mean values of the data are the most reliable. We calculated the joint probability distribution using the statistical properties of the small individual data sets in order to estimate the upper limit risk likelihoods, but the confidence intervals around these estimates can be quite large. The emissions factors and statistical properties of the data distributions are given in Table 4. Emissions factors used by EPA in their report to Congress are also shown for comparative purposes.

Results

Figure 3 shows an example of the joint probability distribution of the capacity/dilution and dioxin/furan data bases. There were some intermediate unit conversion steps involved to generate a concentration versus probability function. The likelihood of any particular inhalation cancer risk level can be derived from the figure by multiplying the predicted concentration by the potency factor. For example, the probability that any future plants built in the United States will show a maximum individual

TABLE 4
COMPARISON OF TRACE POLLUTANT EMISSION FACTORS, WESTON AND U.S. EPA DATA

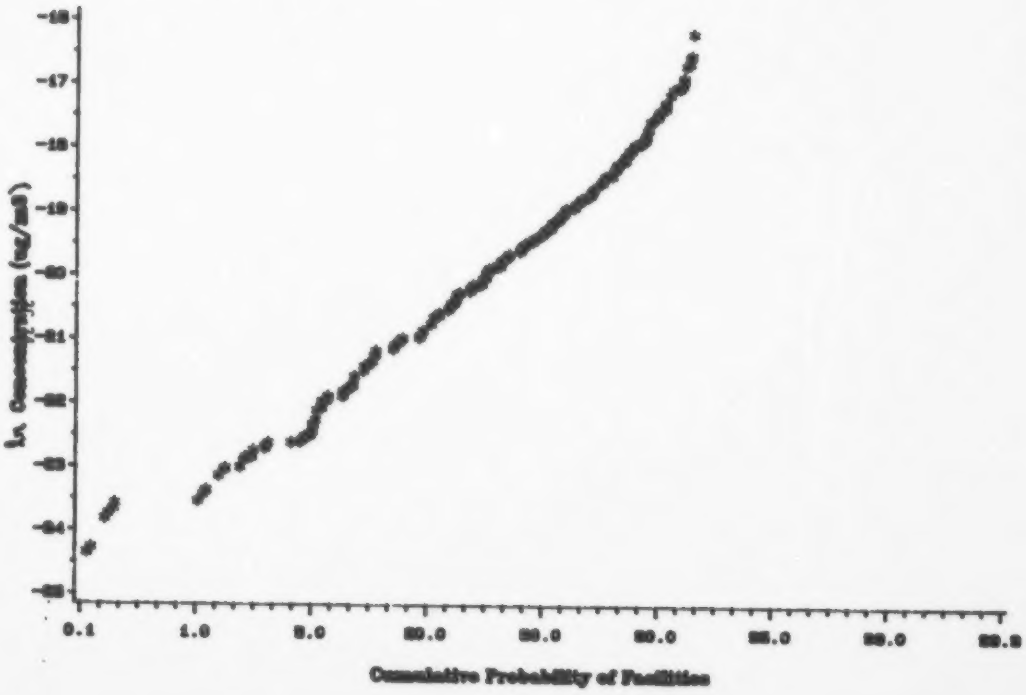
Compound	Emission Factors		
	WESTON		U.S. EPA(1)
	GM	+95%	Range
PCDD/PCDF (ng/m^3)	.96	2.37	12 - 200
PAHs ($ug\ nm^3$)	.17	1.3	10.1
Arsenic (#/ton)	5.9×10^{-7}	2.2×10^{-4}	$4.3 \times 10^{-6} - 2.2 \times 10^{-3}$
Beryllium (#/ton)	7.1×10^{-8}	4.3×10^{-6}	$4.8 \times 10^{-9} - 3.1 \times 10^{-6}$
Cadmium (#/ton)	1.9×10^{-5}	1.7×10^{-4}	$6.0 \times 10^{-5} - 4.8 \times 10^{-3}$
Chrome VI(2) (#/ton)	2.2×10^{-6}	4.0×10^{-5}	$7.1 \times 10^{-6} - 3.4 \times 10^{-4}$
Lead (#/ton)	1.2×10^{-4}	6.9×10^{-4}	$2.4 \times 10^{-4} - 1.5 \times 10^{-1}$
Nickel (#/ton)	5.0×10^{-5}	1.6×10^{-3}	$2.2 \times 10^{-3} - 4.6 \times 10^{-3}$

(1) Transformation of EPA concentration data using 4350 nm^3/ton conversion

(2) Assumes maximum Cr VI/total chrome = 3.5% (Marian Co. observations)

FIGURE 3

Probability of Dioxin/Furans Concentration



inhalation cancer risk due to dioxin/furan exposures of greater than 1 chance per million is less than 0.2 percent, or one chance in 500. The most likely exposure risk is 0.2 chances/million.

This same graphical approach was used to estimate the joint probabilities of inhalation risk for arsenic,cadmium, chrome, nickel, and beryllium. These data are shown in Table 5. The lead exposure distribution was also estimated and compared to the National Ambient Air Quality Standard. The maximum predicted concentration is less than 1/200 of the 1.5 $\mu g/m^3$ NAAQS at the 1/1000 probability.

Conclusions

There is a clear-cut need to develop and use cross-sectional risk assessment models to support regulatory and risk management policies. The results of such modeling are extremely sensitive to the manner in which the input data is handled statistically. In order for the results to be meaningful to the risk management decision maker, the probabilities of the most likely and worst case outcomes need to be clearly stated.

The authors have cross-sectionally modeled the inhalation risks for dioxins/furans and trace metal carcinogens associated with waste-to-energy incineration. It was found that there is a very low probability that the maximum individual cancer exposure risk will exceed one in a million for any of these pollutants. Based on these results, the setting of technology-based standards, e.g., at the lowest observed levels, etc., is not warranted. It is more appropriate to set emission limits on a case-by-case basis using local inputs and conditions as the rationale for regulation. The same technique can be applied to other

TABLE 5
SUMMARY OF CARCINOGENIC RISK
ESTIMATES BASED ON JOINT PROBABILITY RISK MODEL

Compound	Estimated Risk (chances/10^6) $p = 0.5$	$p = 0.01$	$p = 0.0001$	Probability of $1/10^6$
2,3,7,8, T_4CDD T.E.	0.2	0.6	2.0	1/500
Arsenic(1)	0.005	0.031	0.054	>1/10,000
Beryllium	<0.00001	0.0009	0.0055	>1/10,000
Cadmium	0.0016	0.013	0.48	1/6,700
Chrome VI	0.0011	0.07	1.2	1/770
Nickel	0.0011	0.23	1.3	1.670

(1) Based on maximum emissions factor of three observations.

industrial sources providing the modeling and emissions data bases are available.

The risks calculated in this approach are for inhalation risks only. Inhalation risk was the focus of the EPA effort and hence the authors concentrated on the inhalation risks so that meaningful comparisons could be made. With the growing number of multipathway risk assessments on waste-to-energy facilities available at this time, it would be relatively straightforward to apply the same concepts to all pathways.

RFW317

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CONTAMINANT DISPERSION AND DEPOSITION TECHNIQUES FOR USE IN HUMAN EXPOSURE MODELING

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Assessment of health risk from exposure to contaminants emitted from municipal solid waste (MSW) incineration requires estimates of contaminant behavior and transport in the environment. Both dispersion in the atmosphere and deposition to the surface must be quantified before human exposure can be estimated. Typically, health risk assessments have been based upon mathematical modeling of air quality performed to demonstrate compliance with ambient air quality standards (AAQS). Impacts predicted from the ambient air quality modeling analysis have then been incorporated directly into the health risk assessment, either as gases to be inhaled, or(if the model permitted), as particulates to be deposited on soil or water and ingested. The Industrial Source Complex Model (ISC), for example, will output either concentration or deposition. However, a great deal of controversy has developed concerning the accuracy of these models to deal with emissions from resource recovery facilities, especially particulate emissions. Regulatory application of deposition modeling has traditionally focussed on large particles (>20um) associated with coal dust, mining operations, etc. For this reason, experiments developed for determining deposition algorithms used in air quality computer models have generally focussed on larger particles, while the increasing data base on MSW emissions indicates a wide range of particle sizes, with varying percentages in the submicron range. Such small particles have been shown to be deposited at velocities greater than those predicted by simple gravitational settling. In addition, research on MSW emissions points to fine particle enrichment and condensation, increasing the loading of pollutants, especially dioxins, furans, and heavy metals, on particles.

This paper will discuss various techniques for estimating contaminant dispersion and deposition for use in human exposure modeling. Standard techniques, such as applying a deposition velocity to the modelled concentration output, will be reviewed along with more sophisticated methods, such as functional relationships between particle size and deposition velocity (e.g., Sehmel's curves). Approaches used in a selection of health risk assessments and recommended by regulatory agencies will be quantified for a model MSW facility and the sensitivity of the health risk assessment to the various approaches will be evaluated.

AIR EMISSIONS TEST RESULTS FROM THE STANISLAUS COUNTY, CALIFORNIA RESOURCE RECOVERY FACILITY

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ABSTRACT

Ogden Martin Systems, Inc.'s Stanislaus County, California resource recovery facility began construction in August 1986. Commercial operation began in February 1989. This waste-to-energy plant uses Thermal DeNOx to reduce nitrogen oxide emissions to help meet its emission offset requirements under the Clean Air Act. In addition, it is the first large-scale facility in California to pass a rigorous environmental and health risk assessment process with review by the California Air Resources Board (CARB) and the California Department of Health Services (DOHS).

The 800 tons-per-day (TPD) facility comprises two waterwall boiler units (400 TPD each) with dry scrubbers and fabric filter baghouses to reduce acid gases and particulates, respectively. This facility generates approximately 20 megawatts of electricity, 15 of which are sold to Pacific Gas & Electric. Approximately 400,000 residents of the City of Modesto and parts of rural San Joaquin Valley are served by this facility.

Results of the testing for emissions of acid gases, nitrogen oxides, heavy metals, dioxins, and other trace organics are reported.

The original health risk assessment is re-evaluated for cancer risk based on these actual emissions and the revised cancer risk is presented. For the 30-year and 70-year scenarios, the risk level is 11 and 20 times lower, respectively, than levels originally reviewed by CARB and DOHS.

INTRODUCTION

The Stanislaus Waste Energy Company (SWEC), owned and operated by Ogden Martin Systems, Inc., is located in Crows Landing, CA, a rural San Joaquin Valley community near the city of Modesto, CA. This facility is Ogden Martin Systems, Inc.'s first resource recovery project located in California and its first facility using add-on nitrogen oxide emission control equipment. Construction began in August 1986 and was completed in late summer, 1988.

The technical specifications of this plant are shown in Table 1. This plant combusts 800 TPD of municipal solid waste (MSW) in two 400 TPD mass-burn units controlled by the Martin GmbH Stoker Combustion System. Emissions of acid gases, particulates, and nitrogen oxides are controlled with dry scrubbers, fabric filter/baghouses, and Exxon's Thermal DeNOx, respectively.

RESULTS

Air emissions testing for criteria and non-criteria pollutants, heavy metals, dioxins and other trace organics was conducted November 29 through December 10, 1988 by Energy Systems Associates (ESA) of Tustin, CA, an independent testing firm approved by the California Air Resources Board. The schedule of compliance testing and the list of approved test methods are shown in Table 2. Results of the criteria and non-criteria pollutant emissions during the compliance testing program are presented in Tables 3, 4, 5, and 6, and are derived from the data contained in ESA's Report No. ESR 25910-1000 (1). Air emission control efficiencies for nitrogen oxides and sulfur dioxide are shown in Tables 7 and 8. These Thermal DeNO$_x$ control efficiencies are approximate since simultaneous uncontrolled levels are not obtainable with accuracy similar to that of controlled levels, i.e., the uncontrolled levels were determined with the NH_3 shut off prior to the testing with NH_3. In addition, the measured uncontrolled levels are typical of normal fuel nitrogen and do not represent seasonal variation of NO_x levels (2). Based on NO_x removal data shown in Table 7, which reflect a short-term test during only one season with average fuel nitrogen, and on unabated NO_x levels measured at other OMS facilities (2), OMS will not guarantee a NO_x removal rate greater than 35-45%. In addition, there must be allowances for short-term (3-8 hr) NO_x levels up to 225 ppm_{dv} at 12% CO_2 or 7% O_2.

The actual air emissions results are compared with the estimated air emissions used in the October 1986 final Health Risk Assessment done by Radian Corporation (3) for the SWEC facility. The cancer risk is recalculated on the basis of actual, rather than estimated, emission levels and the revised cancer risk is presented in Figures 1 and 2.

ACKNOWLEDGEMENTS

The authors acknowledge the excellent source test effort of Ogden Projects, Inc.'s independent contractor, Energy Systems Associates, and their key test personnel, M. D. McDannel, P.E., and A. C. Bell.

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TABLE 1. TECHNICAL SPECIFICATIONS OF THE STANISLAUS COUNTY RESOURCE RECOVERY FACILITY

Plant Capacity:	
Guaranteed Throughput:	292,000 Tons Per Year
Guaranteed Delivery:	243,300 Tons Per Year
System Description:	
Furnace:	Mass-burning with Waterwall Boilers
Process Units:	Two, 400 TPD per Unit
Stoker Grate:	Martin GmbH, Inclined, Reverse-Reciprocating
Steam Pressure:	865 psig
Steam Temperature:	830°F
Condenser Cooling Medium:	Well Water
Energy Products:	
Net Electricity Generated:	Approximately 15 MW
Customer:	Pacific Gas & Electric Co.
Air Quality Controls:	
Nitrogen Oxide Control:	Exxon's Thermal DeNO$_x$ System
Acid Gas Removal:	Dry Flue Gas Scrubbers
Particulate Removal:	Fabric Filter Baghouses

TABLE 2. SCHEDULE OF EMISSIONS TESTING AT STANISLAUS COUNTY RESOURCE RECOVERY FACILITY

Test No.	1988 Date	Time	Test Method	Unit	Location	Run
01	12/01	0928-1258	U.S. EPA 104	1	Outlet	1
02	12/01	1355-1730	U.S. EPA 104	1	Outlet	2
03	12/01	1850-2225	U.S. EPA 104	1	Outlet	3
04	12/02	0846-1003	U.S. EPA 5, CARB 5	1	Outlet	1
05	12/02	0845-0945	U.S. EPA 8, Modified	1	Outlet	1
06	12/02	0845-0945	U.S. EPA 8, Modified	1	Inlet	1
07	12/02	1130-1247	U.S. EPA 5, CARB 5	1	Outlet	2
08	12/02	1130-1230	U.S. EPA 8, Modified	1	Outlet	2
09	12/02	1130-1230	U.S. EPA 8, Modified	1	Inlet	2
10	12/02	1415-1528	U.S. EPA 5, CARB 5	1	Outlet	3
11	12/02	1415-1515	U.S. EPA 8, Modified	1	Outlet	3
12	12/02	1415-1515	U.S. EPA 8, Modified	1	Inlet	3
13	12/02	1620-1725	U.S. EPA 13B	1	Outlet	1
14	12/03	0825-1055	U.S. EPA 12/101A, CARB 423 & 424	1	Outlet	1
15	12/03	1550-1805	U.S. EPA 12/101A, CARB 423 & 424	1	Outlet	2
16	12/03	1245-1345	U.S. EPA 25A	1	Outlet	1
17	12/03	1443-1553	U.S. EPA 25A	1	Outlet	2
18	12/03	1604-1704	U.S. EPA 25A	1	Outlet	3
19	12/03	1755-1910	U.S. EPA 13B	1	Outlet	2
20	12/04	0903-1536	U.S. EPA Mod. 5, CARB 428	1	Outlet	1
21	12/04	1503-1727	U.S. EPA 12/101A	1	Outlet	3
22	12/04	1655-1837	U.S. EPA 13B	1	Outlet	3
23	12/05	0800-1115	U.S. EPA Mod. 5, CARB 428	1	Outlet	2
24	12/05	1345-1700	CARB 428	1	Outlet	3
25	12/05	1521-1621	U.S. EPA 9, CARB 9	1	Outlet	1
26	12/05- 12/07	2025- -1035	Particle Size	2	Outlet	1

Continued...

TABLE 2. SCHEDULE OF EMISSIONS TESTING AT STANISLAUS COUNTY RESOURCE RECOVERY FACILITY (Continued)

Test No.	1988 Date	Time	Test Method	Unit	Location	Run
27	12/06	0934-1112	U.S. EPA 5, CARB 5	2	Outlet	1
28	12/06	0935-1112	U.S. EPA 8, Modified	2	Outlet	1
29	12/06	0938-1055	U.S. EPA 8, Modified	2	Inlet	1
30	12/06	1217-1334	U.S. EPA 5, CARB 5	2	Outlet	2
31	12/06	1217-1334	U.S. EPA 8, Modified	2	Outlet	2
32	12/06	1220-1335	U.S. EPA 8, Modified	2	Inlet	2
33	12/06	1500-1617	U.S. EPA 5, CARB 5	2	Outlet	3
34	12/06	1500-1617	U.S. EPA 8, Modified	2	Outlet	3
35	12/06	1502-1618	U.S. EPA 8, Modified	2	Inlet	3
36	12/06	1731-1848	U.S. EPA 13B	2	Outlet	1
37	12/07	1353-1633	U.S. EPA 7E, CARB 100	2	Outlet	1*
38	12/07	1647-1927	U.S. EPA 7E, CARB 100	2	Outlet	2*
39	12/07	2000-2240	U.S. EPA 7E, CARB 100	2	Outlet	3*
40	12/08	0020-0300	U.S. EPA 7E, CARB 100	1	Outlet	1*
41	12/08	0410-0800	U.S. EPA 104	2	Outlet	1
42	12/08	0830-1150	U.S. EPA 104	2	Outlet	2
43	12/08	0641-0741	U.S. EPA 7E, 10, CARB 100	2	Outlet	1
44	12/08	0754-0854	U.S. EPA 7E, 10, CARB 100	2	Outlet	2
45	12/08	0904-1004	U.S. EPA 7E, 10, CARB 100	2	Outlet	3
46	12/08	1148-1248	U.S. EPA 25A, CARB 100	2	Outlet	1
47	12/08	1319-1410	U.S. EPA 25A, CARB 100	2	Outlet	2
48	12/08	1534-1634	U.S. EPA 25A, CARB 100	2	Outlet	3
49	12/08	1035-1355	U.S. EPA Mod. 5, CARB 428	2	Outlet	1
50	12/08	1730-2045	U.S. EPA Mod. 5, CARB 428	2	Outlet	2
51	12/08	1325-1545	U.S. EPA 12/101A	2	Outlet	1
52	12/08	2200-0025	U.S. EPA 12/101A	2	Outlet	2
53	12/08	2347-0110	U.S. EPA 13B	2	Outlet	2
54	12/09	0250-0524	U.S. EPA 12/101A	2	Outlet	3
55	12/09	3350-0500	U.S. EPA 13B	2	Outlet	3
56	12/09	0625-0950	U.S. EPA 104	2	Outlet	3
57	12/09	0925-1323	U.S. EPA Mod. 5, CARB 428	2	Outlet	3
58	12/09	1703-1803	U.S. EPA 7E, 10, CARB 100	1	Outlet	1
59	12/09	1811-1911	U.S. EPA 7E, 10, CARB 100	1	Outlet	2
60	12/09	1921-2021	U.S. EPA 7E, 10, CARB 100	1	Outlet	3
61	12/09	2037-2317	U.S. EPA 7E, 10, CARB 100	1	Outlet	2*
62	12/09	2345-0225	U.S. EPA 7E, 10, CARB 100	1	Outlet	3*

*NH$_3$ off

TABLE 3. RESULTS OF EMISSION TESTS FOR CRITERIA AND SELECTED NON-CRITERIA POLLUTANT CONCENTRATIONS FROM THE STANISLAUS COUNTY RESOURCE RECOVERY FACILITY

Parameter	Concentrations at 12% CO_2 Unit 1	Unit 2	Average	Permitted Concentration
Criteria Pollutants				
NO_x (ppm_{dv})	93	113	103	200
SO_2 (ppm_{dv})	2.8	5.3	4.1	30
CO (ppm_{dv})	43	42	43	400
THC (as CH_4, ppm_{dv})	3	4	4	70
Particulate (gr/dscf)				
Total Particulate (including condensible fraction)	0.0108	0.0112	0.0110	0.0275
Solid Particulate	0.0055	0.0022	0.0039	0.011
Solid Particulate of <2 microns	---	0.00004	0.00004	0.008
Selected Non-Criteria Pollutants				
HCl (ppm_{dv})	0.5	2.5	1.28	50
Fluoride (ppm_{dv})	0.24	0.07	0.16	3
NH_3 (ppm_{dv})	3.7	5.0	4.4	50

TABLE 4. RESULTS OF EMISSION TESTS FOR CRITERIA POLLUTANT MASS EMISSION RATES FROM THE STANISLAUS COUNTY RESOURCE RECOVERY FACILITY

Parameter	Unit 1 lbs/day	Unit 2 lbs/day	Both Units lbs/day	Permitted Mass Emission Rate for Both Units lbs/day
NO_x	674	864	1538	2,260
SO_2	31	77	108	800
CO	185	197	382	800
THC (as CH_4)	7	10	17	96
Total Particulate (including condensible fraction)	97	102	199	475

TABLE 5. RESULTS OF EMISSION TESTS FOR CANCER RISK POLLUTANTS FROM THE STANISLAUS COUNTY RESOURCE RECOVERY FACILITY

	Concentrations at 12% CO_2		
Parameter	Unit 1	Unit 2	Average
Total PCDD/PCDF (ng/Nm^3)	6.70	7.13	6.92
Total PAH ($\mu g/Nm^3$)	3.90	2.62	3.26
Total PCB ($\mu g/Nm^3$)	<0.20	<0.18	<0.19
Heavy Metals ($\mu g/Nm^3$)			
Arsenic	0.60	0.93	0.77
Beryllium	<0.00055	<0.00052	<0.00054
Cadmium	1.84	2.35	2.10
Chromium	13.1	10.9	12.0
Nickel	22.5	21.9	22.2

TABLE 6. EMISSIONS OF DIOXINS AND FURANS FROM THE STANISLAUS COUNTY RESOURCE RECOVERY FACILITY IN U.S. EPA TOXIC EQUIVALENTS

Parameter	Unit 1 ng/Nm^3 @ 12% CO_2	U.S. EPA Toxic Equivalents	Unit 2 ng/Nm^3 @ 12% CO_2	U.S. EPA Toxic Equivalents
2378 TCDD	[0.010]	0.000	[0.009]	0.000
Other TCDD	0.386	0.004	0.509	0.005
12378 PCDD	[0.009]	0.000	[0.008]	0.000
Other PCDD	[0.052]	0.000	0.088	0.000
123478 HxCDD	[0.022]	0.000	[0.017]	0.000
123678 HxCDD	[0.018]	0.000	[0.018]	0.000
123789 HxCDD	[0.021]	0.000	[0.018]	0.000
Other HxCDD	0.115	0.000	0.279	0.000
1234678 HpCDD	0.088	0.000	0.087	0.000
Other HpCDD	0.133	0.000	0.065	0.000
OCDD	0.092	0.000	0.068	0.000
2378 TCDF	0.146	0.015	0.152	0.015
Other TCDF	3.186	0.003	3.317	0.003
12378 PCDF	0.165	0.017	0.196	0.020
23478 PCDF	0.091	0.009	0.097	0.010
Other PCDF	1.679	0.002	1.708	0.002
123478 HxCDF	0.176	0.002	0.179	0.002
123678 HxCDF	0.091	0.001	0.076	0.001
234678 HxCDF	[0.013]	0.000	[0.020]	0.000
123789 HxCDF	[0.015]	0.000	[0.012]	0.000
Other HxCDF	0.096	0.000	0.032	0.000
1234678 HpCDF	0.186	0.000	0.154	0.000
1234789 HpCDF	[0.020]	0.000	[0.018]	0.000
Other HpCDF	0.011	0.000	0.014	0.000
OCDF	[0.020]	0.000	[0.018]	0.000
Total Toxic Equivalents		0.053		0.058

Note: Values in brackets are minimum detection limits of not-detectable congeners and are considered as zeroes in calculating averages and toxic equivalencies (4).

TABLE 7. EFFECTIVENESS OF THERMAL $DeNO_x$ IN CONTROLLING EMISSIONS OF NO_x AT THE STANISLAUS COUNTY RESOURCE RECOVERY FACILITY

	Uncontrolled		Controlled		
	NO_x ppm_{dv} @ 12% CO_2	NH_3 lb/hr	NO_x ppm_{dv} @ 12% CO_2	NH_3 lb/hr	Reduction Efficiency %
Unit 1					
Run 1	304	0	85	29	72
Run 2	290	0	99	29	66
Run 3	298	0	95	29	68
Average	297	0	93	29	69
Unit 2					
Run 1	308	0	110	25	64
Run 2	306	0	110	26	64
Run 3	297	0	119	23	60
Average	304	0	113	25	63

TABLE 8. EFFECTIVENESS OF DRY SCRUBBER/FABRIC FILTERS IN CONTROLLING EMISSIONS OF SO_2 AT THE STANISLAUS COUNTY RESOURCE RECOVERY FACILITY

	SO_2		SO_2	
	Inlet	CaO	Outlet	
	ppm_{dv} @ 12% CO_2	lbs per ton MSW	ppm_{dv} @ 12% CO_2	Removal Efficiency %
Unit 1				
Run 1	56.2	18.2	0.3	99.5
Run 2	75.9	18.2	0.8	98.9
Run 3	63.8	16.6	7.3	88.6
Average	65.3	17.6	2.8	95.7
Unit 2				
Run 1	14.7	19.0	2.9	80.3
Run 2	63.3	17.5	5.6	91.2
Run 3	21.4	15.0	7.4	65.4
Average	33.1	17.2	5.3	84.0

Figure 1. Original and Revised 30-Year Cancer Risk

Stanislaus County Resource Recovery Facility

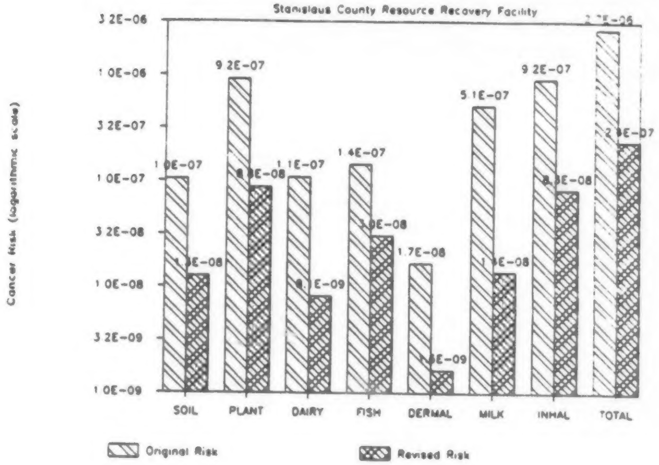
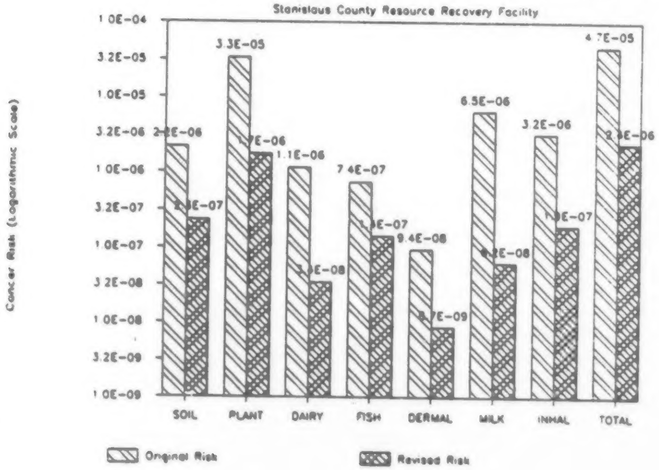


Figure 2. Original and Revised 70-Year Cancer Risk

Stanislaus County Resource Recovery Facility



JACKSON COUNTY (MICHIGAN) RESOURCE RECOVERY FACILITY

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ABSTRACT

The Jackson County waste-to-energy facility has a capacity rating of 200 tons per day (TPD) of municipal solid waste, (MSW). Each of the two 100 TPD unit streams represent current modern day design and operating principles that are directly associated with environmental control.

Each of the unit streams incorporate the Riley-Takuma stoker, combustion and boiler technology. The air quality control system being a dry scrubber and fabric filter supplied by Research-Cottrell.

Included in the facility's performance is compliance with the emission levels imposed by the State of Michigan. Performance trials having been conducted include the trials for environmental control from which the test results show that all emissions were below state limits.

The following text therefrom describes the facility installation and emission control performance.

INTRODUCTION BACKGROUND

Following a comprehensive study the recommendation for the Jackson Facility was initiated in August 1984 by approval of a bond issue by the Jackson County electorate. Subsequent construction began in July 1986 and commercial operation in November 1987. The performance - compliance trials were conducted during April of 1988.

Facility Description

The Jackson Facility has two 100 TPD trains each sized to incinerate municipal solid waste (MSW). The complex having provisions to install a future third train. Commensurate with the facility's needs each unit train has a Riley Stoker supplied Takuma combustion grate system, Riley heat recovery boiler and associated automatic combustion controls. Air quality assurance being further complemented by the dry scrubber and fabric filters (baghouse) supplied by Research Cottrell. Each of the unit trains has an auxiliary gas fired burner situated in the furnace which is employed as necessary during start-up, shut-down and to maintain lower furnace temperatures in excess of 1600°F.

The facility is a waste-to-energy complex employing heat recovery Riley boilers generating superheated steam to a single turbine generator with a gross output of 3.7 MW of electricity. The said electrical power and turbine extraction steam are so used by the adjacent State prison. Excess electrical energy is sold to the Consumers Power Company.

Stoker, Boiler and Combustion Air Systems, (1).

The Riley Stoker system for MSW as shown in Figure 1 employs an integral design of Takuma stoker grate and Riley heat recovery boiler. The boiler incorporates a tubular water wall furnace. Each boiler is designed to generate 24,300 lbs/hr of superheated steam at 615 psig and 650°F.

The stoker grates employed as depicted in Figure 2 have a double hydraulic ram feeder section followed by a reciprocating ignition, combustion and final burn-out grate section. The burning surface is of high grade chromium content cast grate bars.

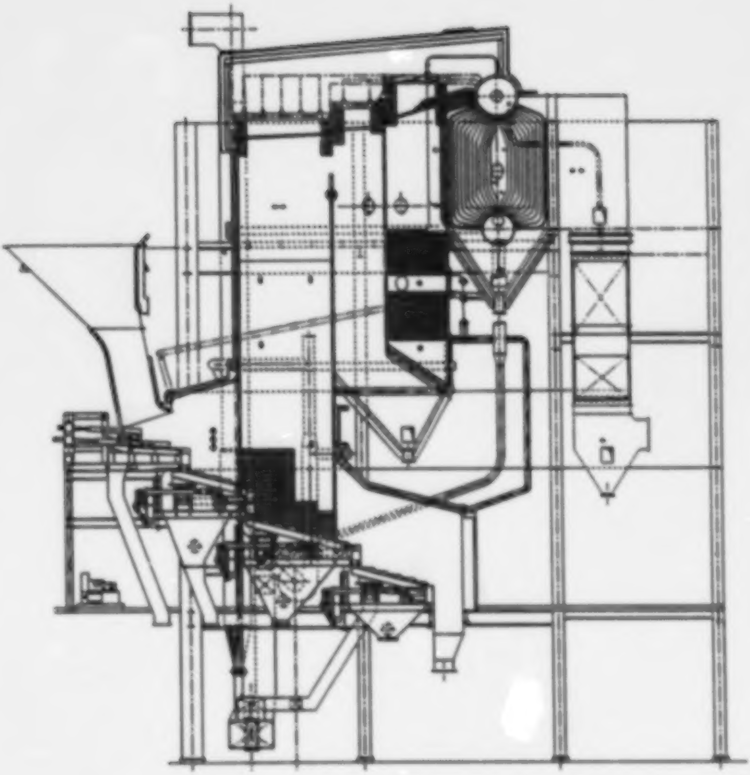
Shown diagrammatically in Figure 3 is the combustion air system. Each unit train has dedicated overfire air and underfire combustion air fans. The overfire air system incorporates staged levels of air injection nozzles. The underfire air supplies air to the ignition, combustion and final burn-out stoker grate section. Steam coil airheaters are so used dependent on the waste quality and heating value.

Combustion Controls

Automatic Combustion Control, (2).

Automatic control of an MSW incinerator, especially the combustion process and boiler, is essential to achieve a constant high quality result. The purpose of automatic combustion control (ACC) is to stabilize combustion and maximize steam generation in line with the plant's main function, which is to dispose of the maximum quantity of waste with the minimum impact to the environment. The Jackson facility employs the Riley - Takuma ACC (microprocessor) system for MSW as shown in Figures 4,5, and 6.

The ACC comprises the following sub-systems:



JACKSON (2 × 100 TPD)

Figure 1 Riley Stoker Corporation Heat Recovery Boiler

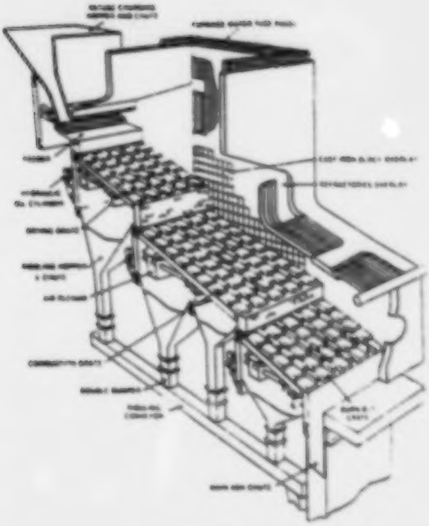


Figure 2 Takuma Boiler Setting

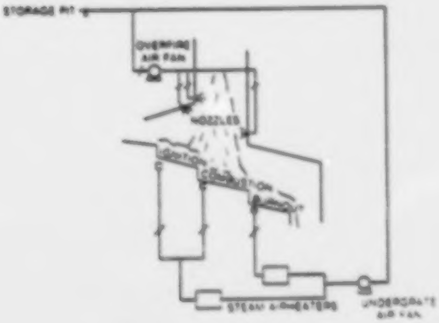

STORAGE PIT
OVERFIRE AIR FAN
NOZZLES
IGNITION
COMBUSTION
BURNOUT
STEAM AIR-HEATERS
UNDERGRATE AIR FAN

Figure 3 Combustion Air System

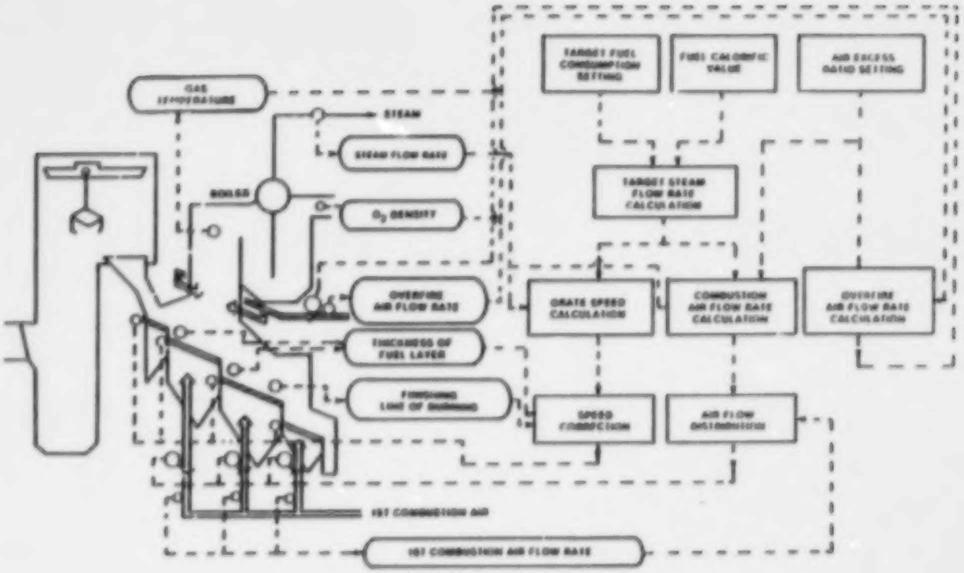

GAS TEMPERATURE
STEAM
STEAM FLOW RATE
BOILER
O_2 DENSITY
OVERFIRE AIR FLOW RATE
THICKNESS OF FUEL LAYER
FINISHING LINE OF BURNING
1ST COMBUSTION AIR
1ST COMBUSTION AIR FLOW RATE
TARGET FUEL CONSUMPTION SETTING
FUEL CALORIFIC VALUE
AIR EXCESS RATIO SETTING
TARGET STEAM FLOW RATE CALCULATION
GRATE SPEED CALCULATION
COMBUSTION AIR FLOW RATE CALCULATION
OVERFIRE AIR FLOW RATE CALCULATION
SPEED CORRECTION
AIR FLOW DISTRIBUTION

Figure 6 Automatic Combustion Control System

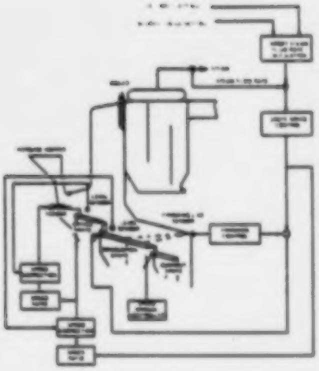


Figure 3 Schematic diagram of grate speed control system

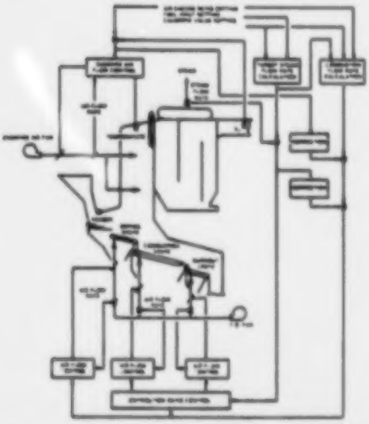


Figure 4 Schematic diagram of combustion air and low oxygen control system

- Grate speed control system
- Combustion air flow control system
- Target refuse throughput control system
- Low oxygen control system
- Furnace outlet flue gas temperature control system

Air Quality Control System. (3).

The air quality control system (AQCS) as shown in Figure 7 consists of a dry scrubber and a fabric filter for control of acid gases and particulate emissions.

The flue gas exiting the boiler enters the quench reactor (dry scrubber) where larger particulates are removed via the cyclonic forces in the inlet section. The gas is then contacted with lime slurry which is atomized into fine droplets by mixing with compressed air in the injection nozzle. The lime slurry neutralizes the acid components and also quenches the flue gas simultaneously.

The quenched gas then flows through a fabric filter (baghouse) for removal of the particulates which include fly ash, acid neutralization products and any unreacted reagent. The clean gas exits the system via the induced draft fan and stack.

Quench Reactor

The quench reactor is the proprietary Research-Cottrell/Teller design with up-flow configuration. It is equipped with a single nozzle for injection of lime slurry which is atomized by compressed air.

Fabric Filter

The fabric filter is pulse-jet design with three modules per train-each of which can be isolated independently for cleaning and maintenance. The bags are made of fiberglass with an acid-resistant coating to protect against acid corrosion. Cleaning of the bags is initiated automatically.

Reagent System

The reagent system consists of a silo for storage of pebble lime (CaO). The lime is fed to the slaker where it is mixed with water to produce a slurry of calcium hydroxide [Ca(OH$_2$)]. A vibrating screen is used to separate the grits and other impurities. The slurry is stored in a storage tank where it is mixed with water to maintain the slurry concentration required for proper operation of the quench reactor.

The lime slurry is injected through the quench reactor nozzle at a controlled rate to achieve the required acid gas removal efficiencies.

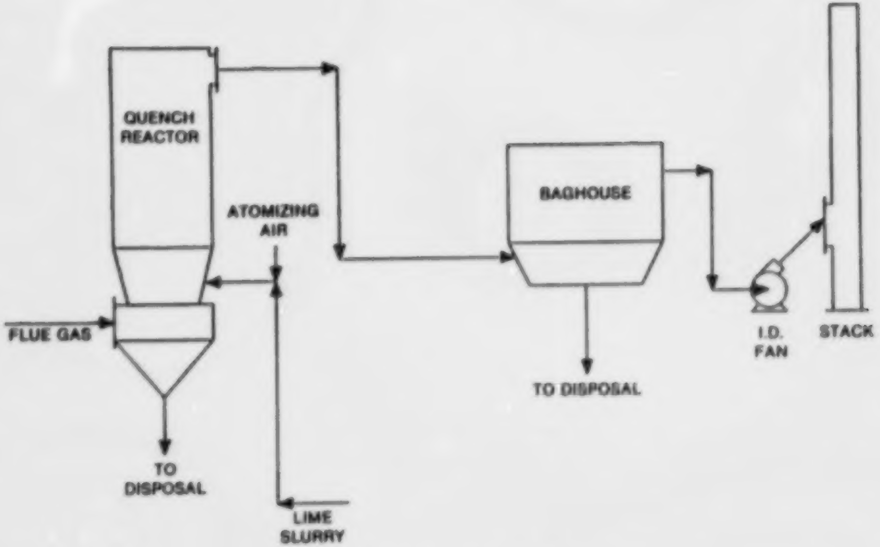

QUENCH REACTOR
ATOMIZING AIR
FLUE GAS
TO DISPOSAL
LIME SLURRY
BAGHOUSE
TO DISPOSAL
I.D. FAN
STACK

Figure 7 Research-Cottrell — Air Quality Control System

COMPLIANCE TESTING

The facility compliance tests were carried out during April 1988. Both unit trains were operating at or near capacity commensurate with the waste heating value. Each unit train stack flue data was obtained for particulate, SO_2, HCl, CO, NO_x, non-methane hydrocarbons, dioxins/furans and certain heavy metals. The test procedures so adopted were generally to established Environmental Protection Agency (EPA) regulations. - sampling methods that appear in Title 40, CFR, Part 60, Appendix A. The respective results of the compliance trials are given in Table 1, (4). The comparison to allowed permitted values being that by review of Table 2. Subsequent discussion thus follows.

Operating Conditions

Both unit trains were firing MSW conducive to generate total steam at an average rate of 48,800 lbs per hour. The overfire air quantity was approximately 30 percent of the total air, the underfire air being heated by the steam coil airheaters to approximately 200°F in the combustion zone and 150°F in the burnout zone.

The furnace exit gas temperatures were maintained in the region of 1200-1300°F being commensurate with calculated lower furnace conditions in the order of 1700-1800°F. The auxiliary burner(s) were not in operation.

The stack-flue gas temperatures were an approximate mean of 300°F with O_2 concentrations in the order of 9.8 percent (dry volume %) for unit 1 and 9.0 percent for unit 2. The evolved stack flue gas flow's were 16500 actual cubic feet (acfm) from unit 1 and 15500 acfm from unit 2, based on average conditions.

The recorded low gas temperatures in the furnace's are considered due to relatively little furnace fouling/slaging resulting in a high heat transfer efficiency.

The scrubber and fabric filter system incorporating the lime slurry controls were operational, maintaining the stack flue temperatures. The range of oxygen content in the stack flue gas being considered typical of such a facility design and operational practices.

Nitrogen Oxides (NOx) Carbon Dioxide (CO) and Total Hydrocarbons

The maximum allowed NOx at Jackson is 5.43 lbs/ton of MSW which relates to 311 ppmv at 12% CO_2 dry gas. The average data for both units was 154 ppmv being less than half than that allowed. The associated CO average was 30 ppmv at 12% CO_2 with actual gas conditions on a consistent basis.

The total hydrocarbons weighted as methane (CH_4) were consistently below 2 ppmv for (unit 1) and 4.7 ppmv for unit 2. This relates to an average of 0.3 mg/dscm (-0.2 ppmv) and 0.05 mg/dscm (0.6 ppmv) for Units 1 and 2 respectively.

TABLE 1 JACKSON COUNTY FACILITY COMPLIANCE TRIAL RESULTS

Emission (a)	Unit 1. (b)	Unit 2. (b)
NOx ppmv	158	150
CO ppmv	30	30
THC mg/dscm, (d)	0.30	0.05
TSP gr/dscf, (c)	0.00147	0.00127
SO_2 ppmv	4.37	3.23
HCl ppmv	22.0	17.0
Arsenic µg/dscm	12.43	8.36
Cadmium µg/dscm	12.70	10.46
Chromium µg/dscm	9.05	3.59
PCDD + PCDF, Expressed as 2, 3, 7, 8 TCDD Equivalents µg/dscm	4.96×10^{-5}	2.67×10^{-5}

Notes

a. Values given are based on 12% CO2 dry gas.

b. All data are average values at full load operation.

c. TSP (particulates) front catch only, gr/dscf being grains per dry standard cubic feet.

d. THC: Total Hydrocarbons weighted as methane, data shown being less than 1 ppmv equivalents.

The consistency of emission control by combustion technology is an essential operational feature. Shown typically in Table 3 is an example of the continuous emission monitoring data. In all cases the NOx, CO and THC measured emissions were well below the allowed-permitted level providing evidence that the combustion technology and automatic controls are operating in a consistent and efficient manner.

Total Particulate (TSP)

Particulate control is maintained by the throw-out chambers within the boiler geometry and ultimately controlled by the fabric filter (baghouse).

TABLE 2 JACKSON COUNTY

Allowable Permit Emissions;	State of Michigan, Dept. of Natural Resources.
NOx (1)	311 ppmv (relates to 5.43 lbs/Ton of MSW)
CO (2)	400, 113 and 63 ppmv
TSP	0.015 grains/dscf
SO_2	86 ppmv or 50% reduction efficiency
HCl	105 ppmv or 90% reduction efficiency
PCDD	0.52 ng/NM^3 3 hour average
PCDF	0.50 ng/NM^3 3 hour average

Notes

1. NOx based on 3 hour average
2. CO time averages are; one hour, 24 hours and annual, respectively.
3. ppmv and grains/dscf corrected to 12% CO2 dry gas.

Although the allowed emission rate of TSP is 0.015 grains per dry standard cubic feet of gas corrected to 12% CO_2 the measured data was some ten times below this limit.

The measured data was significantly lower than design ranging from 0.0018 grains down to 0.001 grains per dry standard cubic foot corrected to 12% CO_2. The average concentrations being 0.00147 gr/dscf for Unit 1 and 0.00127 gr/dscf for Unit 2 corrected to 12% CO_2. Throughout the trials all three baghouse modules were in service on both unit streams.

Acid Gases (SO_2 and HCl)

During the trials inlet loading values of SO_2 and HCl were not measured thus collection efficiencies cannot be advised. However, earlier trials did compute that for HCl and SO_2 that collection efficiencies in the order of 95 percent can be anticipated, (3). The compliance test results relative to absolute stack values averaged 19.50 ppmv (12% CO_2) for HCl and 3.8 ppmv (12% CO_2) for SO_2, far less than the allowed emission limits.

TABLE 3 CONTINUOUS EMISSION MONITOR DATA
JACKSON COUNTY RESOURCE RECOVERY FACILITY
JACKSON COUNTY, MICHIGAN
APRIL 27, 1988 UNIT 2, RUN 2

Time	O2 (%)	CO2 (%)	SO2 (ppm)	NOx (ppm)	CO (ppm)	HC (ppm)
10:13 AM	9.8	10.0	-1.8	120	41	-0.4
10:14 AM	10.4	9.8	-1.8	105	46	-0.2
10:15 AM	11.0	9.9	19.5	114	46	0.2
10:16 AM	9.8	10.0	-4.5	126	42	0.5
10:17 AM	9.8	9.1	-4.5	117	42	0.4
10:18 AM	10.4	9.4	3.5	109	46	-0.2
10:19 AM	11.0	9.0	-4.5	117	44	-0.4
10:20 AM	10.4	9.2	-4.5	123	44	1.1
10:21 AM	11.0	9.6	11.5	126	37	1.4
10:22 AM	9.8	9.9	-4.5	114	46	0.0
10:23 AM	9.1	10.0	-4.5	138	44	-0.4
10:24 AM	8.5	11.0	4.8	153	30	1.7
10:25 AM	8.5	10.4	-1.8	149	27	-0.4
10:26 AM	10.4	9.4	-3.1	137	26	-0.3
10:27 AM	10.4	9.5	-4.5	148	19	-0.4
10:28 AM	9.8	10.0	11.5	142	23	0.9
10:29 AM	9.8	9.9	0.9	143	24	0.5
10:30 AM	9.8	10.2	-4.5	144	14	-0.4
10:31 AM	9.8	10.4	8.9	146	14	1.7
10:32 AM	9.8	10.6	-4.5	148	12	-0.4
10:33 AM	9.1	10.0	-4.5	133	14	-0.4
10:34 AM	9.1	10.3	10.2	144	14	1.1
10:35 AM	9.1	11.1	-4.5	154	14	-0.4
10:36 AM	8.5	11.1	-4.5	149	15	-0.4
10:37 AM	9.1	11.0	8.9	140	16	1.4
10:38 AM	7.8	11.0	-4.5	125	20	1.5
10:39 AM	9.1	10.7	-4.5	115	18	1.0
10:40 AM	9.1	11.2	-4.5	141	16	-0.4
Average	9.5	10.2	-0.5	125	32	0.1
AVERAGE @ 12% CO2			-0.6	147	37	0.1

Heavy Metals

Conducive to the incineration process and fuel related influences of MSW the contaminants of arsenic, cadmium and chromium were recorded. Associated with the particulate capture and resulting stack emission levels as measured should be duly considered as part of the history book and data bank on such pollutants. It is considered that the recorded values of Arsenic, Cadmium and Chromium are typical of the low emissions expected from solid waste incineration facilities.

Dioxins and Furans

The total dioxin and furan emission levels were of focal interest being recorded at 4.96×10^{-2} and 2.67×10^{-2} ng/dscm for Units 1 and 2 respectively. These levels are part of the lowest values of such pollutants recorded by modern facilities.

Concluding Remarks

The Jackson Facility can be credited with full compliance with the atmospheric - environmental restrictions so imposed by the State of Michigan. These state requirements are not considered lenient as they were imposed by appropriate facility permitting - licensing laws and regulation a credit to such procedures. The Riley Stoker and Takuma equipment design at Jackson has demonstrated that by state-of-the-art technology that emission control of NOx, CO and THC is a factual operating feature of the facility.

The incorporation of dry scrubbers and fabric filters ensuring acid gas and particulate emission control. Each system, ie combustion technology and the application of scrubber-fabric filters thus provide an integrated atmospheric control system.

The emission levels so indicated being below that authorized by the State permitting authorities provides the confidence that the technology has the controls to maintain environmental compliance even when considering the variable characteristics of municipal solid waste.

*

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PERFORMANCE TRENDS FOR THE NEW BALTIMORE SOUTHWEST FACILITY: PROJECTIONS AND REALITIES

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ABSTRACT

A standard terminology has been developed for describing the performance of refuse power plants (RPP's). Major parameters such as availability, utilization, load factors and capacity factors have been defined for major equipment and systems. For cogenerating RPP's, total energy recovery efficiencies have been derivated. Subsequently this standard terminology has been applied to the Saugus and Baltimore RPP's where the first served as the reference plant for the second. Similarities in the behavior of both plants are discussed. For the Baltimore RPP trendlines show a decrease in waste processing capacity which may be related to changes in the wastestream. Finally, increasing steam exports are responsible for a continued upward trend for winter peaks in overall energy recovery efficiency.

BACKGROUND AND INTRODUCTION

During 1981, the Northeast Maryland Waste Disposal Authority negotiated with Wheelabrator-Frye Inc. for construction of the new Baltimore Southwest Facility (BSWF). This was a full service procurement for a facility capable of processing up to 2,250 $STPD_7$ (short tons per day on a 7-day a week basis) and generating up to 60 MW of electrical power.

Since the BSWF was to replace the failed Monsanto pyrolysis plant in downtown Baltimore, only proven technology was acceptable. Wheelabrator-Frye intended to apply the experience gained from its successful waterwall mass burning plant in Saugus. In fact, it formally declared Saugus as the reference plant for its proposal to the Authority.

The Authority's team of technical and financial consultants was charged with the responsibility to assess the performance of the Saugus plant. A key task was to judge the relevance of the Saugus data to the Baltimore performance guarantees being negotiated.

After the award, Wheelabrator built the BSWF and performed the acceptance tests. The plant started full-scale commercial operation on May 1, 1985 without a hitch. Since that time, the BSWF has nearly completed four years of continuous operations, exceeding performance projections.

It is the purpose of this paper to develop standard terminology which lends itself to the analysis of plant performance. This terminology will then be successfully applied to four different sets of data which are characteristic of the following:

(1) Operating results of the older reference plant which has reached maturity.

(2) Performance guarantees and test criteria which were agreed upon during negotiations for the new plant.

(3) Test results from the new plant which confirm that construction contract requirements have been met.

(4) Operating results from the new plant which demonstrate that operating contract requirements have been satisfied.

Based on comparative analysis, the questions "how relevant was the reference plant" and "how appropriate were contract requirements for the new plant" will be answered.

I. PLANT PERFORMANCE CRITERIA

It is hardly possible to find a single criterion by which RPP performance can be judged. In fact, it is difficult to correlate all the parameters which characterize plant performance. Given the twofold purpose of the RPP, that it is to dispose of the largest possible amount of refuse and to produce the greatest possible amount of energy, one can establish at least two sets of criteria.

The first of these are the acceptance test criteria which include the minimum amount of refuse processed and the minimum amount of energy produced. The first can be stated as the minimum tonnage processed (MTP). The latter is often expressed as the minimum specific electrical generation rate (MSEGR) in [kWh produced/ST refuse processed] which must be attained by the RPP in order to be accepted. Depending on whether [kWh produced] or [kWh exported] is involved, one can refer either to the Gross MSEGR or to the Net MSEGR. The acceptance test criteria are usually agreed upon in the RPP Construction Contract. The test period may be 24 h, 168 h or even 672 h.

The second set of criteria deals with the performance guarantees which the operator makes as part of the RPP Operations Contract. In the case of a full service procurement, the test criteria and the performance guarantees are produced by the same corporate entity.

Again, the MTP, the Gross MSEGR and the Net MSEGR are involved except that these criteria refer to a much longer period of time, i.e. usually 1 year or 8,760 h. Customarily, because of the longer time period involved, the performance criteria are set lower than the test criteria. The margin between the two is a negotiated item which involves ultimately a judgement call, although the historical performance record of a reference plant is often used as a guide.

II. PERFORMANCE MONITORING AND REPORTING

In properly managed RPP's, a continuous log is kept by the operating crews. Entries are made either in 15 minute or 1 hour intervals. Such parameters as the refuse firing rate, the steam production rate and the electrical generation rate are entered. In addition, for each major piece of equipment the start-up and shut-down times are recorded. Also, the cause for each shut-down is noted, be it equipment failure or refuse shortage.

The hourly entries are then totaled and reported on daily summary sheets. Daily totals are then transferred to weekly and monthly performance reports. Besides being reviewed by the crew chiefs and the chief operating engineer, the monthly performance reports are then sent on to senior plant management.

Depending on the reporting format agreed upon in the operating contract, monthly, quarterly and/or annual reports may be forwarded to the owner and the monitoring agency.

Since many plants nowadays feature central control rooms with microprocessors and a full complement of sensors and recorders, it may be possible to replace the handwritten logs either in part or in entirety with electronically generated data. By means of automatic data logging it may be possible to produce even monthly reports by EDP.

However, it may still be desirable (in order to keep operators alert) or even necessary (in order to obtain legally acceptable hard copies) to have some of the data-taking done by hand.

Before discussing data analysis, it is necessary to agree upon a standard terminology which can be readily understood.

III. PROPOSED STANDARD TERMINOLOGY

In discussions with Authority staff, fellow consultants, and representatives of the financial and industrial communities, we have frequently encountered a lack of standard terminology when evaluating key performance parameters of RPP's. These parameters include, but are not limited to, availability, capacity, utilization and the like.

In order to avoid misunderstandings and to facilitate performance analysis, we have developed the standard terminology which is discussed below. (A summary of this terminology is contained in Table 1.)

1. Availability

The one term most frequently used is that of "plant availability," but in connection with continuously operating waste disposal and resource recovery plants, such terminology is of only limited use. "Equipment availability" would be a far more meaningful expression, especially if it is related to specific pieces of equipment which are the building blocks of any processing system.

"Being available" simply means that a particular equipment is able to run and perform within its design specifications. For example, an electric motor is "available" when it can respond to the flick of a switch and run. Consequently, "availability" means the number of hours during which a particular piece of equipment is available for operation. Usually it is the number of available hours divided by the total number of hours in the reference period of time used. In the case of a 1-year reference period, an availability of 0.800, or 80%, would mean that a particular piece of equipment must be able to run 0.800 x 24 x 365 = 7,008 hours.

Whether or not this equipment will actually run for 7,008 hours is another story. For example, it may be the operator's choice to assign a perfectly operable boiler to standby duty. This is often done in European plants where the operator prefers to have one unit always in reserve. (Note: In some instances this choice is dictated by the fact that raw waste bypassing is forbidden.) It also happens when there is a temporary shortage of refuse. Thus in a certain 3-unit plant, even when all equipment is available, two units may be in operation and one unit may be in reserve or on standby. A unit which is undergoing inspection, repair, modification or replacement is obviously not available to operate and perform its assigned function.

We can now define availability for an individual piece of equipment, or for a string of several sequential pieces of equipment which must perform in unison in order to make a complete unit work:

$$A_i = \frac{\text{available hours of equipment during reporting hours}}{\text{total hours during reporting period}}$$

By examining the operating log, the operator is able to calculate out unit availabilities on a monthly basis. If the plant is equipped with multiple, parallel and identical units, then the monthly report will tabulate out unit availabilities, such as A_1, A_2, A_3 ... A_n for all units installed.

In a plant designed for waste-to-energy conversion, i.e. a refuse power plant or RPP, a complete chute-to-stack system can be subdivided by function into several major subsystems, each of which has a unit availability of its own. See also Figure 1. Such subsystems include the furnace system, the boiler system, the residue removal, and the air pollution control system, etc. Within a given processing line, various unit availabilities may affect one another. For example, if a boiler blows a tube this does not necessarily mean that the grate underneath it is inoperable as well. In other words, the boiler may be unavailable when the grate is still available, and vice versa.

Generally, it is unlikely, if not impossible, to run one subsystem, or a specific unit, within a particular chute-to-stack system without running all the others. Therefore, only a single availability is determined for the entire chute-to-stack system in order to retain a measure of simplicity. This is done regardless of which subsystem was at fault.

For reasons which go beyond the scope of this paper, unit availability and equipment reliability are closely interrelated. Let it suffice to note that reliability is an essential tool used by the plant designer, whereas availability is an essential tool by the plant operator.

During reliability analysis, however, individual units and their individual components must be examined in order to determine potential failure modes and effects (FMEA's). Such reliability analysis during the design phase is important to improved availability during the operating phase. The idea is to identify by analysis the weakest link in any unit or system and then upgrade its design so that a more uniform on-line performance of all pieces can be accomplished.

For a multi-system plant, average availability can be determined on an annual basis as follows:

$$A_{i-n} = \frac{hai + \ldots han}{n \text{ x } 24 \text{ x } 365} = \frac{hai + \ldots han}{n \text{ x } 8{,}760} \;;$$

wherein "ha" represents the total hours during the course of the year during which a particular chute-to-stack system is available. "n" represents the number of identical and parallel systems.

Average annual availability for the Saugus RPP, for example, would be based on two chute-to-stack systems, and it can be expressed:

$$A_{1,2} = \frac{ha1 + ha2}{2\text{x}24\text{x}365} = \frac{ha1 + ha2}{17{,}520}$$

Similarly, average annual availability for the Baltimore RPP would be based on three chute-to-stack systems, or:

$$A_{1,2,3} = \frac{ha1 + ha2 + ha3}{3\text{x}24\text{x}365} = \frac{ha1 + ha2 + ha3}{17{,}250}$$

While annual averages are convenient for describing performance trends of older plants with a long-term data base, newer plants need to be analyzed on a monthly and/or quarter-annual basis in order to spot abnormalities.

Furthermore, when predicting future performance based on historical data, the confidence on such predictions is raised by resorting to larger sample sizes, i.e. the more frequent monthly rather than the fewer yearly observations.

Accordingly, the denominator in the above formulae will be changed to 1,440 h for average monthly and to 4,320 h for average quarterly data in the case of Saugus. For Baltimore which features 3 chute-to-stack systems, the denominator will change to 2,160 and 6,480 h.

Obviously, it is the goal of all professional plant operators to achieve high availability. However, by definition, availability can only approach unity, but it cannot exceed it. The closer to unity, the better the rating for a given plant, but we need to remember that high availability comes as the end product of good design, construction and operation practices. Thus it begins with plant design.

2. Utilization

The derivation of utilization is similar to the foregoing discussion of availability, except that utilization is closer to operational reality. Equipment utilization is the amount of time that a particular equipment is being used. Thus a piece of equipment may be available, but it cannot be used because there is a disruption in the supply of refuse. It could also be that the operator decides to hold a particular piece of equipment in reserve. This latter approach is a familiar strategy used in many European plants. There are also reasons as to why available equipment is not being used.

Because of equipment interdependency, it is propitious to talk again in terms of chute-to-stack systems utilization.

For a multi-system plant, average utilization can be defined on an annual basis as follows:

$$U_{1-n} = \frac{hui + \ldots + hun}{n \times 8,760}$$

wherein "hu" represents the total hours during the course of a year during which a particular chute-to-stack system was in use.

It is common for well-run RPP's to push utilization as close as possible to availability. Ideally speaking, utilization should equal availability. Our own observation in many RPP's indicates a typical difference of at least 2-3%. Start-up and shut-down procedures alone tend to account for this difference.

As mentioned before, any unit which is in operable condition, but which happens to be shut down because of a refuse shortage, should be counted as being "available." Consequently, availability increases over utilization. On the other hand, a unit which is in operation may break down sooner because of increased wear and tear, thus leading to a further decrease in both availability and utilization. It is difficult to quantify these second order effects and they are merely mentioned here for discussion purposes.

3. Load Factors

Nameplate capacity, or maximum continuous rating (MCR), is a statement from the manufacturer which describes the full load rate at which a specific equipment, unit or system can be safely operated on a continuous basis.

In RPP's this would mean the rate at which a grate can burn refuse, or the rate at which a boiler can generate steam.

The load factor indicates at what fraction of nameplate capacity, the equipment is actually being used at any particular point in time. Under ideal operating conditions, the load factor would reach unity and continue at unity as long as the equipment stands up.

Investment costs relate directly to the size, or the capacity, of an installed system. Therefore, it is in the investor's interest to see to it that equipment is run at full capacity all of the time in order to maximize revenues. In practice, this is not possible for a variety of reasons, among which we would cite inconsistencies in the composition of refuse, especially varying degrees of moisture and inerts, content, feeder problems, irregularities in energy use profile, personnel difficulties, instrumentation and control malfunctions, and the like.

As a result, recordings of load factors resemble the behavior of human body temperature which besides undergoing instantaneous and diurnal variations also recognizes abnormal levels, such as during incidence of fever or freezing.

In chute-to-stack systems, there are two major load factors which we need to recognize: the grate load factor, or LFg, and the boiler load factor, or LF_b. Both of these factors are related to each other, but once again it would go beyond the scope of this paper to enter into a discussion of their complex interrelationship. Ideally speaking, these load factors would be nearly identical, but it has been our experience in auditing a number of RPP's that the boiler load factor is generally lower than the grate load factor. Often, this is by a substantial amount. To some extent, this is due to the fact that a grate system tends to be more forgiving than its related boiler. Its response time may be in the order of 15-30 minutes compared to a few minutes in boilers.

Average load factors can be determined for each unit on a monthly, quarterly or annual basis. We would suggest the following definition:

For a particular grate:

$$LF_{gi} = \frac{\underline{\text{tons of refuse processed during reporting period}}}{\text{hui x Cgd}}$$

Typically the tons of refuse processed are weighed by a load cell attached to the feeding system and are recorded, or totalized, by the plant's EDP system. The hourly processing capacity, or Cgd, is available from the manufacturer's nameplate. Usually it is verified during the plant's acceptance test. Traditionally, it is expressed in short tons per hour (stph).

For a particular boiler:

$$LF_{bi} = \frac{\underline{\text{pounds of steam generated during reporting period}}}{\text{hui x Cbd}}$$

The pounds of steam generated are usually recorded by a steam flow meter or totalizer. This is a very vital record, because energy revenues earned are primarily based on this documentation. In addition, strip chart recorders provide a continuous record of instantaneous steam flow. By integrating the area under the curve, the same result should be gotten. Strip charts are most helpful to the operator in permitting him to adjust the plant's output control system.

The hourly steaming rate, or Cbd, should also be stated on the boiler's nameplate and it should be verified during acceptance tests. It is usually expressed in pounds per hour (lbs/hr).

For a plant with multiple but identical systems, an average load factor can be calculated on an annual basis:

For the grates, we can write:

$$\overline{LFg}_{i-n} = \frac{\underline{\text{tons of refuse processed during reporting period}}}{\text{(hui + ... + hun) x Cgd}}$$

For example, during 1980, the Saugus RPP reported a

$$\overline{LF}_{g1,2} = \frac{374,555}{(6,213 + 6,765) \times 31.25} = \underline{0.924}$$

which is quite respectable even by European standards. See Ref. (1).

For the boilers we can write:

$$\overline{LFb}_{i-n} = \frac{2,489,343 \times 10^3 \times 0.685}{(6,213 + 6,765) \times 157,250} = \underline{0.836}$$

(Note: 0.685 is a steam enthalpy correction factor which was used by WFI to allow for low pressure steam returned by GE for in-plant use at the Saugus RPP.)

Both load factors, as derived from Saugus data, are respectable, but neither of them are unity. While such load factors are recognized by operators as detractors from annual performance goals, they are generally ignored by proposal makers.

During our more than 15 years of involvement with resource recovery, we have rarely encountered an LFg which has approached or exceeded unity. However, there have been exceptions. For example, in the Kiel plant in West Germany, unity was exceeded during some of the early years. This practice was discontinued when the overloaded grates were generating more flyash than the the APC system could handle. See Ref. (2).

In Saugus, during one of the early years (1977), the $\overline{LF}$g reached unity once, which seemed to indicate either that low calorific refuse was fired, or that the grates were overloaded, or both. The practice of overloading was discontinued, which is not surprising, in view of the grate failures experienced at Saugus.

With regard to the boilers, the LFb is always bound to remain below unity. In fact, we do not know of a single instance, where unity has been exceeded. This is because being a pressure vessel and sensitive to changes from many variables, a refuse-fired boiler can never generate steam at a perfectly constant output level. Instead, major steam parameters such as pressure, temperature and flow rate fluctuate on an instantaneous, an hourly and a daily level. Any irregularities on the part of the energy user tend to send back a ripple effect which may further aggravate a serious and complex control problem. While pressure and temperature can be held with modern control systems to within ±2% about average, steam flow is much more difficult to control.

4. Plant Capacity Factors

The annual plant capacity factor, or PCF, is frequently used in the electrical utility industry in order to relate generated capacity to installed capacity. Utility companies are required by law to report PCF data to the Public Service

Commission.

The PCF approach of reporting bypasses the foregoing discussion of availability, utilization and load factors. Without commenting on the many causes and effects which may detract from a plant's performance, the PCF is simply the bottom line approach which presents the year's final result in a single figure.

The North American Electric Reliability Council (NAERC) publishes statistics on the performance of electrical power plants according to capacity and type of fuel. See Ref. (3)

The category which comes closest to refuse power plants are coal power plants in the 1-99 MW capacity range. For the 1983-87 observation period, from the data we have extracted the following category averages:

AF = 0.844 ; LF = 0.705
UF = 0.526 ; PCF = 0.371

It is obvious from these statistics that a large amount of installed coal-fired power plant capacity remains unused.

In a refuse-fueled power plant, conditions are more difficult. Refuse is a non-conforming fuel which frequently changes its composition. However, more important is the fact that a refuse-fueled plant must perform two separate, but equally important tasks: waste disposal and energy recovery. Consequently, these plants produce two major revenue streams of nearly equal proportions, as well as tipping fees and energy sales. (Note: Other potential revenue streams such as scrap sales or interest earned by contingency funds are disregarded in the context of this paper.)

Both revenue streams relate directly to the PCF and, given the fact that installed refuse power capacity costs considerably more than coal power capacity, the RPP operator must accomplish a much higher PCF. Since the PCF is essentially the product of the UF and the LF, the RPP operator must push both much higher than those references for the coal power plants. For example, in the U.S. today, well-run RPP's routinely achieve UFG = 0.89 and LFG = 0.95. The resultant product, or the PCFG = 0.85, has already been exceeded by some of the best plants, such as the Baltimore RPP.

In accordance with these two disparate functions, it appears logical to classify annual plant performance of RPP's into two distinctly different parts, i.e., the plant capacity factor of the grates, PCFg, and the plant capacity factor of the boilers, PCFb.

Using mathematical notations, we can write:

For the grates:

$$PCFg = \frac{\text{Annual Waste Tonnage Processed}}{n \times 8{,}760 \times Cgd}$$

wherein "n" = the number of grates installed and "Cgd" = hourly unit waste processing capacity in stph at full load, or MCR.

For the boilers:

$$PCFb = \frac{\text{Annual Pounds of Steam Generated}}{n \times 8{,}760 \times Cbd}$$

wherein "n" = the number of boilers installed and "Cbd" = hourly unit steaming rate in lb/hr at full load, or MCR.

We had previously surveyed a number of older European plants and we found in a sample of 21 plants in West Germany during the 1960 to 1975 period an average PCFg = 0.518. In the Saugus plant size range, the best of the German plants achieved a PCFg = 0.671 and a PCFb = 0.594 (over an observation period of 11 years). See Ref. (4).

Interestingly enough, Saugus managed to attain a PCFg = 0.642 and a PCFb = 0.589 (over an observation period of 5 years). Neither in the German plant nor at Saugus did the PCFb ever equal the PCFg. Under optimal conditions, the two factors may come within a few percentage points of each other, a fact which further attests to the difficulties associated with the use of refuse as a boiler fuel.

5. Specific Steaming Rate

In modern refuse power plants, grates and boilers are designed as well-integrated combinations. Both grate and boiler must have mass and heat transfer capacities which are well matched so that only a minimum goes unused in either one.

Obviously there are limits to the range of refuse composition which any grate boiler combination can accommodate. These limitations are best expressed in the so-called furnace diagram. For a given design, the furnace diagram indicates that if refuse has a low calorific value, the grate tends to be the limiting part. Conversely, if refuse has a high calorific value, the boiler tends to be the limiting part.

In Europe, where the calorific value of refuse experienced a dramatic increase during the seventies, heat transfer limitations of the older boiler designs forced a reduction of the annual waste processed. In many cases this was a painful experience, because reduced throughput meant reduced earnings from tipping fees. Although energy costs increased, they could not totally compensate for the loss in tipping fees.

In the U.S., some of the plants built in the late seventies or early eighties are experiencing similar problems. In recent years, it has been observed that community recycling efforts have actually helped to further raise the calorific value of refuse. Thus the SSRa was driven towards the SSRd, leading to the need to reduce the rate of firing refuse.

As a result, the newer plants designed in the late seventies featured relatively large boilers. This trend can be best expressed by the design specific steaming rate, or SSR. As specified by the design engineer:

$$\text{SSRd} = \frac{\text{hourly steaming rate}}{\text{hourly waste processing rate} \times 2{,}000} = \frac{\text{Cbd}}{\text{Cgd} \times 2{,}000}$$

wherein "Cbd" = hourly unit steaming rate in lb/hr, "Cbd" = hourly unit waste processing rate in stph, and 2,000 lb = 1 ST.

In the sixties, European plants were being designed for SSRd = 1.5 to 2.0. In the seventies the range was more 2.0 to 3.0, and several of the most modern were designed for 3.00 to 3.30. It should be noted, however, that these figures refer to the generation of superheated steam. For saturated steam, these figures tend to be proportionately higher because of the thermodynamics involved.

Under actual operating conditions, the SSRd becomes a goal which is not often reached or maintained. Therefore, an actual specific steaming rate can be defined as:

$$\text{SSRa} = \frac{\text{Annual Pounds of Steam Generated}}{\text{Annual Tons of Waste Processed} \times 2{,}000}$$

For example, before its boilers were reconfigured, the Duesseldorf RPP in West Germany managed to post SSRa = 2.07 for a SSRd = 2.09 during 1980. See Ref. (4). The Saugus plant achieved a SSRa = 2.36 for a SSRd = 2.52 (after boiler modifications and derating from SSRd = 2.96.

6. Performance of Turbine Generator Set

In a true RPP such as Baltimore, the performance of the turbine generator (TG) set is equally important to that of the chute-to-stack system. The same standard terminology previously developed for the grate-boiler combination can also be applied to the TG with some minor modification.

We can write for the average annual utilization factor:

$$UFTG = \frac{hai + \ldots + han}{n \times 8{,}760}$$

We can write for the average annual load factor:

$$LFTG = \frac{Ei + \ldots + En}{nx(hai + \ldots + han)xP}$$

We can define the various terms as follows:

hai = aggregate operating hours per unit and year

n = number of turbine generator sets installed

Ei = aggregate electrical generation per unit and year in [MWh/yr]

P = unit power generation rate (@ 100 % MCR) in [MW]

Since Baltimore features only a single TG with 60 [MW] of installed capacity, we can simplify the above formulae:

$$UFTG = \frac{ho}{8{,}760} \times 100\% \quad ; \quad LFTG = \frac{E}{ho \times 60} 100\%$$

And by combining the annual plant performance factor referred to the TG can be determined as

$$PCFTG = UFTG \times LFTG$$

Depending on what degree of sensitivity is desired for the analysis, monthly and quarterly factors can be determined simply by substituting the appropriate number of hours.

7. Total Energy Recovery Efficiency

In the case of the Baltimore RPP which as a true cogenerating power plant exports both electricty and steam, the total energy recovery efficiency is also of significance. Depending on whether or not in-plant energy consumptions is included or excluded, we can speak of a gross or not total energy recovery efficiency.

In order to derive a formula for gross total energy recovery efficiency, we need to make several assumptions:

(a) The higher heating value for the refuse fired is known from the "living calorimeter method" and has proved to be fairly constant. In the case of the Baltimore RPP, we set HHV ≈ 5,000 Btu/Lb constant.

(b) Small amounts of electricity which are purchased during brief periods of turbine outages, are added to the amount of electricity normally generated for station power.

(c) Station power is useful energy, because most of it is used for incineration, i.e. fulfillment of the waste disposal obligation.

(d) For thermal export, a net enthalpy of 1,389 Btu/Lb steam is stipulated at the point of delivery to the district heating network when the condensate return is taken at zero. For conversion from thermal to equivalent electrical energy, the equivalent of 1kWh ≙ 3,414 Btu is applied.

We can now write the following equations:

$$\mathrm{MWh}_{th}\ \text{input} = \frac{\text{(ST) MSW processed} \times 2{,}000\ \text{(Lb/ST)} \times 5{,}000\ \text{(Btu/Lb)}}{3{,}414\ [\text{Btu/kWh}] \times 1{,}000\ [\text{kWh/MWh}]}$$

or

$$\mathrm{MWh}_{th}\ \text{input} = [\text{ST}]\ \text{MSW processed} \times 2.929\ [\text{MWh/ST}]$$

$$\mathrm{MWh}_{th}\ \text{output} = \frac{\text{(Lb) Steam Exported} \times 1{,}389\ \text{(Btu/Lb)}}{3{,}414\ [\text{Btu/kWh}] \times 1{,}000\ [\text{kWh/MWh}]}$$

or

$$\mathrm{MWh}_{th}\ \text{output} = [\text{Lb}]\ \text{Steam Exported} \times 0.407 \times 10^{-3}\ [\text{MWh/Lb}]$$

$$\mathrm{MWh}_{el}\ \text{output} =$$

Finally, total gross energy recovery efficiency can be defined:

$$\eta\ \text{Gross} = \frac{\mathrm{MWh}_{el}\ \text{output} + \mathrm{MWh}_{th}\ \text{output}}{\mathrm{MWh}_{th}\ \text{input}} \times 100\%$$

IV. APPLICATION OF STANDARD TERMINOLOGY TO SAUGUS AND BALTIMORE RPP's

A series of figures were prepared which show performance trends in the older Saugus reference RPP and the newer Baltimore RPP.

1. Chute-to-Stack Systems in Saugus RPP

Figure 2 shows the performance of the chute-to-stack systems as expressed by the UFG1,2, the PCFG1,2 and the LFG1,2. A major drop observed during 1979 can be attributed to modifications which were made to the grate boiler systems. During more recent years, all three parameters leveled off, indicating that operations have reached maturity.

2. Chute-to-Stack Systems in Baltimore RPP

Figure 3 shows the UFG1-3*, PCFG1-3* and LFG1-3* trends. Because of the shorter data base available, quarterly rather than yearly averages were plotted. As a result, more peaks and valleys are visible. One very large dip early in 1987 was due to a refuse shortage caused by heavy snowfall. (Note: Baltimore's refuse collection trucks were pressed into snow removal service.) While some leveling off occurred during the more recent quarters, it should be noted, however, that the PCFG* shows a downward trend. For additional data, see Ref. (5).

3. Baltimore and Saugus Comparison

In Figure 4, the PCFG*'s for Baltimore and Saugus are superimposed. A certain cyclic behavior is indicated for both. Since the period is approximately one year in most instances, either seasonal changes in the refuse stream (availability and/or composition) or recurring maintenance events are the underlying causes. More research with the plant operators would be required in order to make a better determination.

It should be mentioned that the PCFG*'s as graphed are not the absolute values. Instead these are actual values divided by the performance goal for each plant. In the case of Saugus the goal was to reach, or exceed, an average PCFG*=0.80, while for Baltimore an average PCFG*=0.85, or better, was desired.

4. Declining Capacity at Baltimore

In Figure 5, the PCFG* for Baltimore appears to be declining as a function of operating time. In order to check further on this trend, the same trend line is shown but on a more expanded scale. With the help of linear regression analysis, a downward trendline was calculated. Since the correlation factor reached the 80% level, significance should be attached to this observation. If this trend should persist, the annual performance guarantee for tonnage throughput may be in trouble.

Although the exact reason has not been determined, it is possible that the higher heating value (HHV) of Baltimore refuse has increased at a faster rate than expected. Thus, an adjustment of the performance guarantee may be required. (Note: A reference HHV=4,500 Btu/Lb was used during the procurement process.) The fact that the UFG* does not show a downward trend seems to further strengthen the hypothesis that increases in the HHV are the source of the problem.

5. Performance of Turbine Generator Set in Baltimore RPP

Using again the standard terminology discussed in Section III before, Figure 6 was prepared in order to track performance of the turbine generator set, or TG. A high degree of reliability in this subsystem is evidenced by the UFTG being consistently above 0.99. The only exception, was the inspection and maintenance during the early part of 1987.

The LFTG* and the PCFTG* track each other rather closely. Both show a cyclic behavior which is a reflection on energy marketing. Basically, the period is again one year. Since the RPP exports as much steam as possible during the winter, both the LFTG* and PCFTG* fall off rather sharply. Conversely, during the summer when steam exports are at a minimum both reach their peak. The trend is towards further increased steam exports, although this trend does not appear to be a strong one.

6. Overall Energy Recovery by Baltimore RPP

With the definitions derived in Section III, and the assumptions as stated, Figure 7 was drawn up. During the early months when the RPP was exporting only electricity, the overall energy recovery efficiency ranged from about 18 to 23%. With the beginning of steam exports in 1986, there has been a progressive increase in the winter peaks, specifically the peaks have increased from 38 to 46 and to 49%. From the more detailed monthly reports it can been seen that increased steam exports are responsible for the advantageous development. See also Ref. (5). On the other hand, the summer valleys appear to be holding steady at about 20%.

The component for electrical in-plant consumption has remained remarkably stable, ranging from 2% to 3%. In the future, electrical in-plant consumption is expected to increase if acid gas scrubbers are added.

CONCLUSIONS & RECOMMENDATIONS

It is hoped that the methodology developed can be applied to other RPP's beyond Baltimore. As shown by its application to the Saugus and Baltimore examples, it lends itself to detecting significant changes in RPP operations. The attainment of mature operations, the seasonality of energy recovery efficiency and the decrease in throughput capacity were promptly discovered. The high degree of equipment utilization which has been achieved at Baltimore is probably the optimum possible anywhere. It certainly is more than what was expected from any comparison with the reference plant. Thus far only basic statistical programs have been applied. The work with newer and more advanced programs is still in progress. Therefore, a qualitative statement regarding the closeness of predicted performance to historical performance cannot be made as yet.

Assuming that the decline in processing capacity at Baltimore was indeed caused by increases in the energy content of refuse, there appears to be one message: rather overdesign than underdesign refuse-fired boilers. With the advent of increased recycling, this need may become stronger yet.

In order to cope better with such developments, we would like to recommend formation of an industry-wide forum of operators. After adoption and perhaps refinement of the standard terminology proposed here, a more meaningful information exchange may take place in the future.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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Table 1. DEFINITION OF MAJOR OPERATING PARAMETERS FOR REFUSE POWER PLANTS(a)(b)

Parameter	Abbreviation	Formula	Units
Plant Capacity Factor-Grates	PCFG*	$\frac{W^*}{n \times h^*c \times Cgh} \times 100$	%
Plant Capacity Factor-Grates	PCFG*	$\frac{LFG^* \times UFG^*}{100}$	%
Average Load Factor-Grates	LFG*	$\frac{W^*}{h^*o \times Cgh} \times 100$	%
Average Equipment Utilization-Grates	UFG*	$\frac{h^*o}{n \times h^*c} \times 100$	%
Waste Tonnage Processed by Plant	W*	$W^*1+W^*2+W^*3$	ST
Number of Installed Grates	n	-	-
Calendar Time	h*c	-	h
Unit Grate Processing Capacity (@100% MCR)(c)	Cgh	-	stph
Aggregate Operating Time	h*o	$h^*o1+h^*o2+h^*03$	h
Plant Capacity Factor-Turbogenerator	PCFTG*	$\frac{E^*}{n \times h^*c \times P} \times 100$	%
Plant Capacity Factor-Turbogenerator	PCFTG*	$\frac{LFTG^* \times UTG^*}{100}$	%
Average Load Factor-Turbogenerator	LFTG*	$\frac{E^*}{h^*o \times P} \times 100$	%
Average Equipment Utilization-Turbogenerator	UTG*	$\frac{h^*o}{n \times h^*c} \times 100$	%
Electrical Generation by Plant	E*	-	KWh
Number of Installed Turbogenerators	n	-	-
Unit Power Generation Rate (@100% MCR)(c)	P	-	KW

Notes: (a) Depending on the observation period used, the plain acronym stands for annual averages, the acronym with a dot stands for quarterly averages and the acronym with an asterisk stands for monthly averages.

(b) Similar definitions can be used for annual observation periods.

(c) 100% MCR stands for 100% of Maximum Continuous Rating.

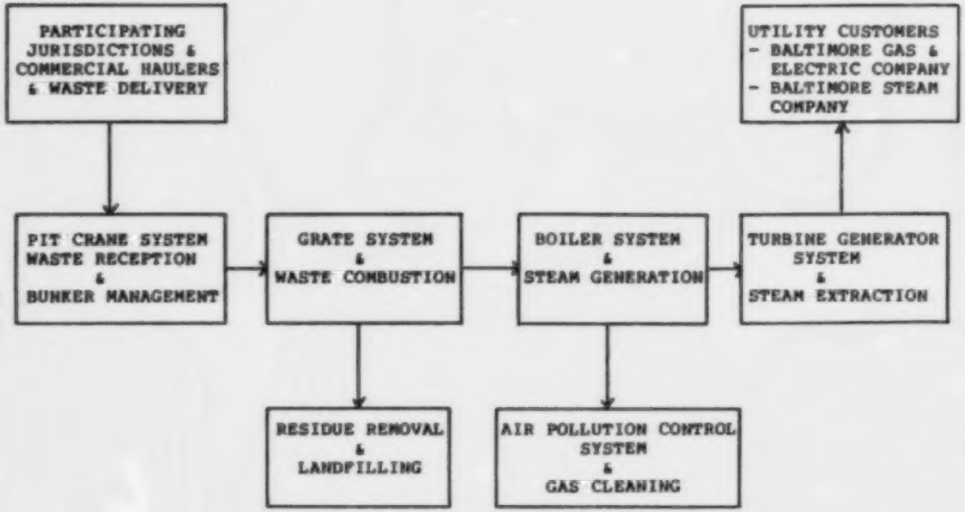
PARTICIPATING JURISDICTIONS & COMMERCIAL HAULERS & WASTE DELIVERY
PIT CRANE SYSTEM WASTE RECEPTION & BUNKER MANAGEMENT
GRATE SYSTEM & WASTE COMBUSTION
BOILER SYSTEM & STEAM GENERATION
TURBINE GENERATOR SYSTEM & STEAM EXTRACTION
UTILITY CUSTOMERS
- BALTIMORE GAS & ELECTRIC COMPANY
- BALTIMORE STEAM COMPANY
RESIDUE REMOVAL & LANDFILLING
AIR POLLUTION CONTROL SYSTEM & GAS CLEANING

Figure 1. SUBSYSTEMS IN A TYPICAL CHUTE-TO-STACK SYSTEM

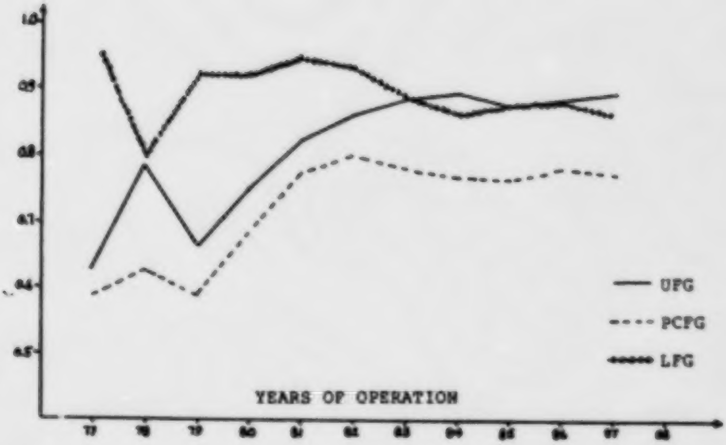

Figure 2. ANNUAL PERFORMANCE AVERAGES OF SAUGUS REFERENCE RPP

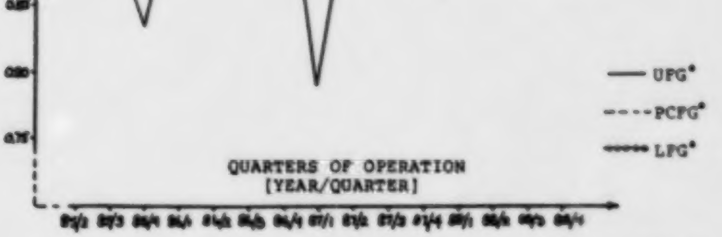

Figure 3. QUARTERLY PERFORMANCE AVERAGES OF BALTIMORE RPP

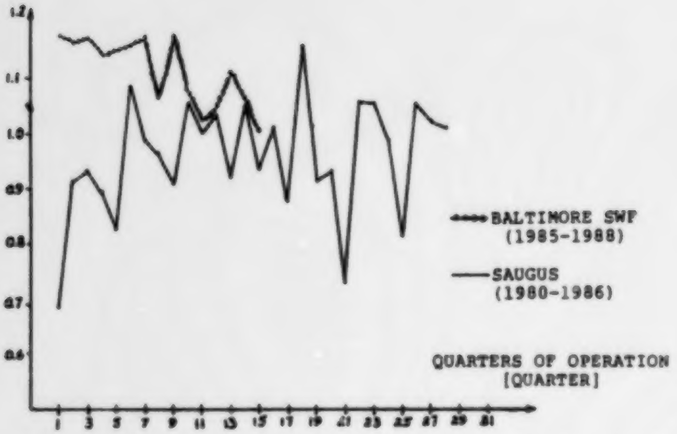

Figure 4. ADJUSTED QUARTERLY PCPG®'s FOR BALTIMORE AND SAUGUS RPP's

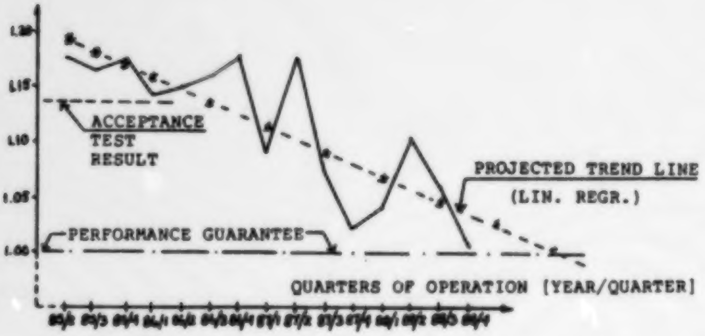

Figure 5. TREND OF PCPG® FOR BALTIMORE RPP (Quarterly averages, adjusted for performance guarantee)

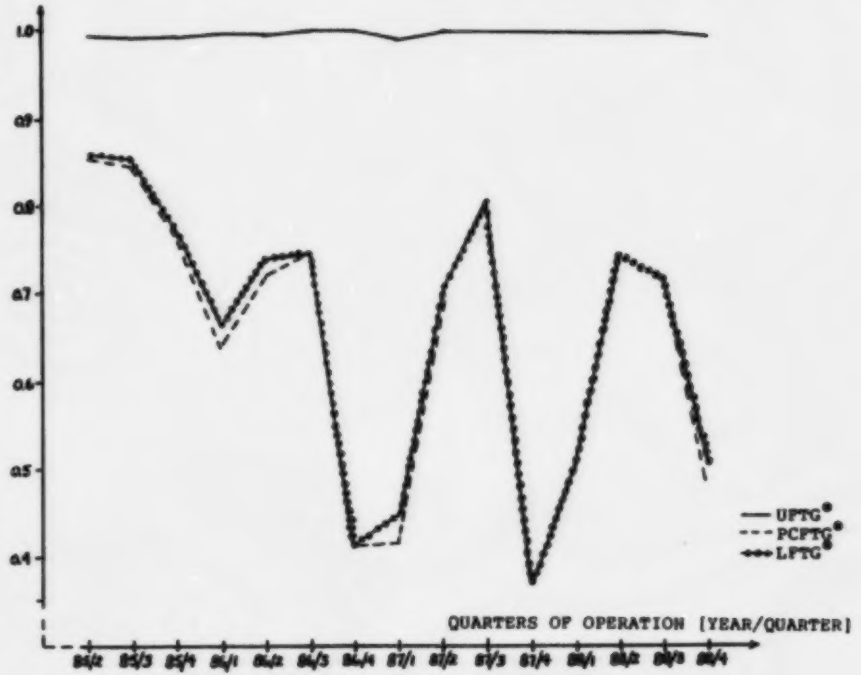

Figure 6. QUARTERLY PERFORMANCE AVERAGES FOR TURBINE GENERATOR SET IN BALTIMORE RPP

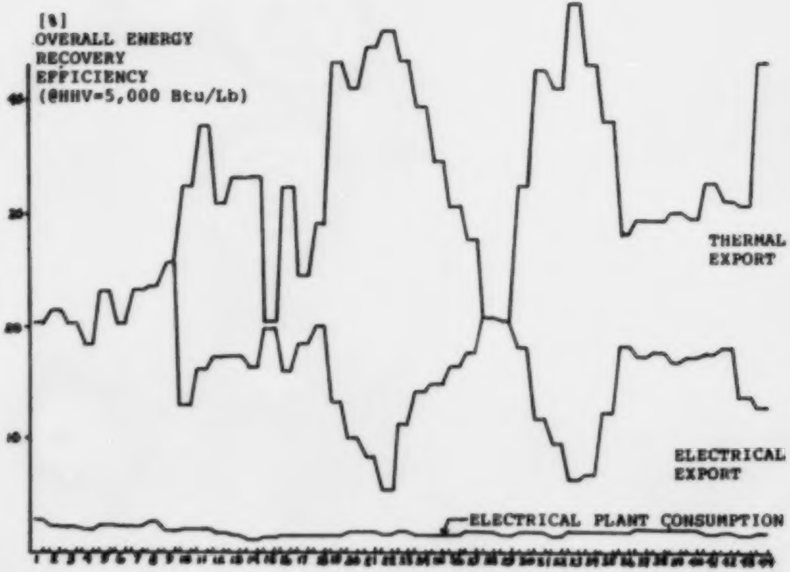

Figure 7. ENERGY RECOVERY TRENDS IN BALTIMORE RPP

LABORATORY PILOT SCALE EVALUATION OF LEAD RECOVERY FROM MUNICIPAL SOLID WASTE INCINERATOR ASH

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ABSTRACT

High levels of potentially leachable toxic metals such as Pb, Cd, and Cr, in the ash residues require consideration of treatment technologies prior to disposal or utilization of the ashes in other applications. Extraction studies revealed information on the leaching characteristics of the ashes and on the ability of the extraction solutions to separate these metals from the ash matrix. The kinetics of the Pb, Cd, and Cr extraction were determined through a series of batch extraction studies. The specific incinerator design and the equilibrium conditions strongly influenced the ash matrix and subsequently the extraction kinetics. Recovery of the metals from the waste extract solution was accomplished utilizing electrochemical plating techniques. A laboratory scale pilot plant was designed to continuously treat ash residues at a rate of 1 kg/hr through extraction and electrochemical techniques. Initial pilot plant operational study results are presented.

INTRODUCTION

Due to decreasing available landfill space, the minimization of the amount of disposable solid materials is imperative. The incineration of municipal solid waste (MSW) results in a 75% reduction in volume from the disposal of residual ashes, as compared to MSW disposal [1]. Heavy metal concentrations extracted from MSW ashes, however, were found to exceed the Extraction Procedure Toxicity test (EP Tox) limits [2]. The EP Tox test is the procedure established by the United States Environmental Protection Agency (USEPA) in testing for leachable inorganic species from solid wastes. The leachability of heavy metals from MSW ash residues must be considered, prior to disposal or utilization of these materials.

Extraction studies employing ash obtained from several resource recovery facilities provided insight to the leaching characteristics of the ashes and the dependency on the extractant solution composition [3,4]. The kinetics of the leaching process were defined and the electrochemical recovery of the metals in solution was investigated [5]. These studies led to the design and operational parameters of a laboratory scale pilot plant, incorporating continuous extraction, separation, and electrochemical processes. The results of the initial pilot plant operation experiments are presented.

Incinerator Designs

Two incinerators were sampled for ash residues, one located in Canada and one in Massachusetts. The Canadian incinerator includes a three tiered vibrating grate primary combustion chamber and a secondary combustion chamber. Solids are fed to the combustion chamber using a hopper and have about a three hour residence period. Sixty percent of the combustion air is provided as underfire air, the remainder of the air is introduced above the burner beds. A full waterwall is provided in the incinerator. A shell and tube heat exchanger cools down the exit gases and fly ash, and separation of the gases and the fly ash occurs in an electrostatic precipitator.

The Massachusetts incinerator utilizes a baffled, three tiered combustion chamber in which the solids are periodically moved tier to tier via a hydraulic ram. The solid residue is quenched in the ash pit and then removed, while the gases and the fly ash pass through a secondary combustion chamber. The ash is separated from the combustion gases utilizing a charged gravel bed and a bag house. Detailed descriptions of both incinerators are available [4].

Kinetics Studies

A series of batch extraction studies investigated the ability of the extractant solution to separate and remove the metals from the ash matrix [5]. Various ash residues were investigated, the Canadian fly ash (CF3) results will be presented. The extractant solution utilized was 1.0 N NaCl (aqueous), acidified with HCl to achieve equilibrium pH of 3.0. This solution was found to be the most efficient in earlier experimentation [4]. The extraction was carried out at a ratio of 1 gram of ash to 20 ml of extractant. Separation of the solid and liquid phases is carried out through vacuum filtration. The waste extract was analyzed measured for heavy metal (Pb, Cd, Cr) concentration via atomic absorption, pH, and conductivity, and their interdependence investigated. The leaching process was investigated at several time periods (in triplicate) up to a steady state time period.

The kinetics of the leaching process for CF3 ash was investigated through observation of the time dependence of the measured variables. The CF3 waste extract pH increased from an initial extractant pH of 0.41 to a steady state equilibrium pH of 2.83 (Figure 1). The lead extraction curve for CF3 peaked at a reaction time of 10 minutes, and subsequently decreased (Figure 2). Concentration levels in the extract at this time period were 69.1 mg Pb/L, or 1380 ug Pb/g ash. The lead extraction process

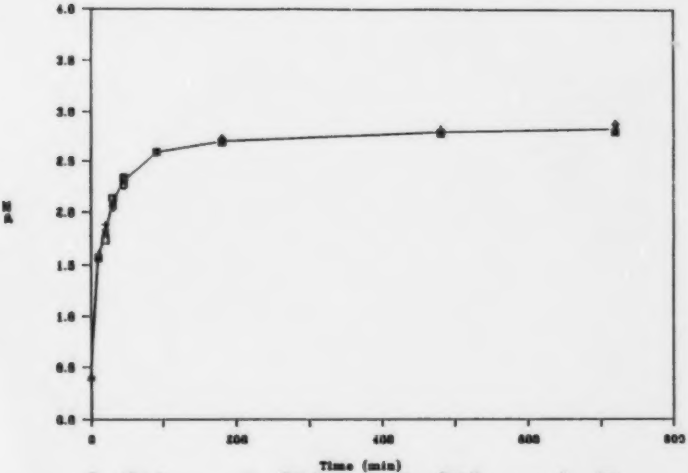


Figure 1 - CF3 Batch Kinetics Studies, pH response in extract

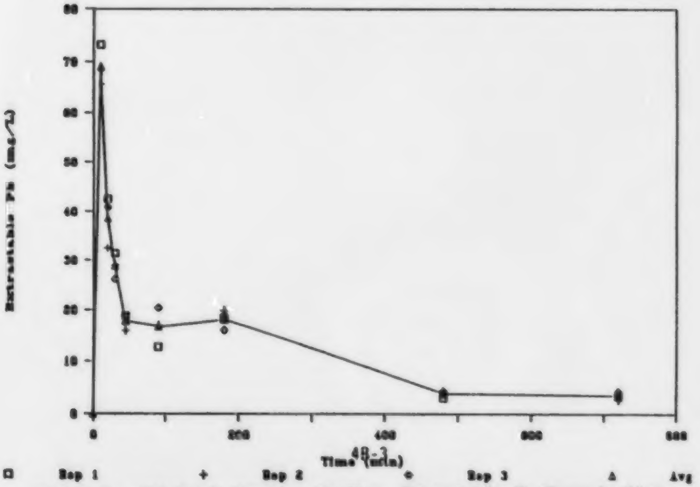


Figure 2 - CF3 Batch Kinetics Studies, Extractable Pb Concentration

for CF3 incinerator ash was observed to be dependent on the pH of the system, upon comparison of the lead and pH curves for the ash. The rate of removal of lead occurred at a much faster rate than the rate of the neutralization process for the ash.

Electrochemical Recovery

Electrochemical methods offered the advantage of the recovery of a relatively pure metal in a usable form; whereas conventional precipitation methods generate a flocculent which requires disposal. Electrodeposition of metals does not require the addition of any extraneous compounds to an already complex solution.

A parallel-plate electrochemical cell was designed for lead recovery. Lead was used as the cathode in this cell. Cyclic voltammetry was carried out using this design on standards of known metal concentration and compared to the aqueous extracts in order to determine the operational parameters of the lead recovery process. The current response observed in the cell utilizing a 180 mg Pb/l standard solution indicated a plating potential of -0.6 V for lead in this solution. Subsequent cyclic runs performed on CF3 extract at a pH of 3.00 and a lead concentration of 175 mg/l exhibited a peak at this value, indicating that lead may be plated out of the extract solution.

In order to determine the amount of metal recovered, timed studies were performed on the lead standard in a CF3 extractant solution. Lead concentration was measured as a function of time, as was the current (Figures 3,4). Both the current and the lead concentration showed the same response. This is indicative of the fact that the amount of lead recovered was controlled by the current in the cell and reached a lower limiting concentration of 110 mg Pb/l.

PILOT PLANT DESIGN AND OPERATION

A laboratory-scale pilot plant was designed to continuously treat MSW incinerator ash residues; refer to Drawing 1 for a simplified process flow diagram. The ash is fed to the continuous stirred tank reactor (CSTR) via a feed hopper/screw pump system at a rate of 1 kg/hr. Extractant solution is fed to to the CSTR at a rate of 20 kg/hr; this flow rate maintains the 1 gram to 20 ml extraction ratio. A pH controller system is utilized to maintain an setpoint pH within the CSTR; this optimum pH value was previously determined from the results of the kinetics studies. The volume in the CSTR is the controlling parameter for the residence time of the solid-liquid reaction. The liquid separation operates continuously with recovery of both phases, utilizing a 10 micron nylon membrane. The treated solid ash residues are stored in a catch basin for further experimentation. The waste extractant is pumped through a basket filter for further filtration and then sent through the electrochemical cell and then recycled. Electrochemical plating of the lead in solution allows for recovery of the metal in a relatively pure form; this process occurs in a multiplated cell equipped with polymer coated titanium anodes (commercial product) and lead cathodes. The majority of the process equipment in contact with the solid

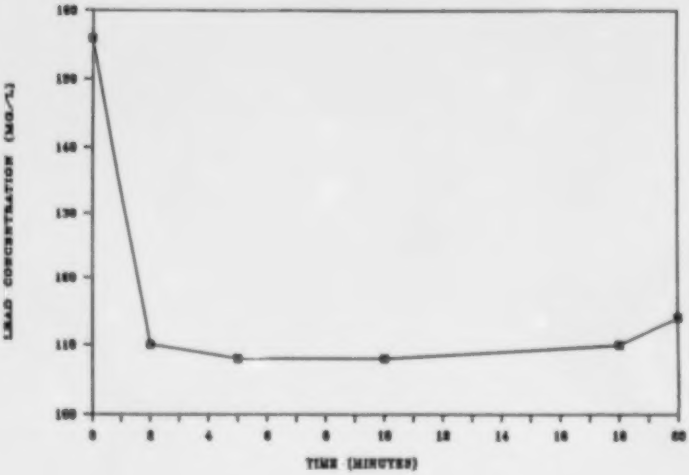


Figure 3 - CF3 Extract Batch Plating, Fb Response

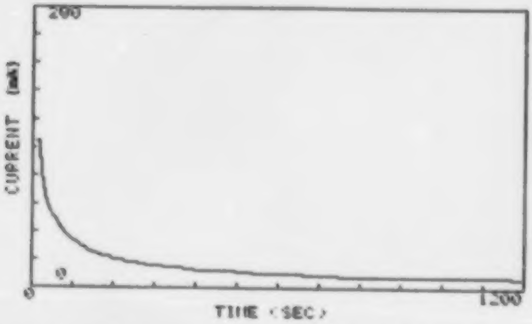
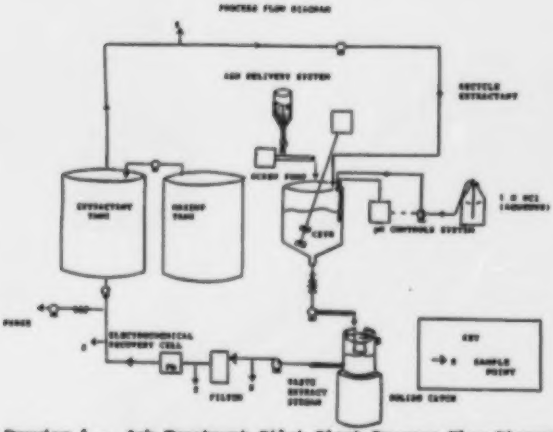


Figure 4 - CF3 Extract Batch Plating, Current Response


PROCESS FLOW DIAGRAM
ASH DELIVERY SYSTEM
RECYCLE EXTRACTANT
EXTRACTANT TANK
SCREW FEED
CSTR
pH CONTROLS SYSTEM
PURGE
ELECTROCHEMICAL RECOVERY CELL
FILTER
WASTE EXTRACT STREAM
SOLIDS CATCH
KEY
SAMPLE POINT

Drawing 1 - Ash Treatment Pilot Plant Process Flow Diagram

or liquid phases are constructed out of inert plastic materials to avoid interference with the metals analyses as well as avoid corrosion due to the acidic nature of the process.

Extraction Process

Initial investigations were carried out to test the extraction portion of the pilot plant, excluding the recycle stream. The CF3 incinerator ash residues were utilized in these experiments. The pH controller setpoint was set at a pH of 1.59 to control the extraction reaction at the optimum level. This optimal level occurs at the peak of lead extraction from the ash (refer to Figures 1 and 2 for pH and lead concentration curves). The pilot plant was operated for 2 hours and waste extract samples were obtained prior to the electrochemical portion. Subsequent atomic absorption analyses of the waste extract stream revealed high levels of extractable lead in solution, matching the concentration levels at the optimum conditions in batch studies (Table 1).

Subsamples of the treated CF3 ash were obtained and assayed for total metal content. One gram aquilots were digested with 20 ml each of nitric and perchloric acids [6]. Solids were removed using vacuum filtration, and the filtrate was quantitatively diluted to 100 ml and analyzed on an atomic absorption spectrophotometer for metals species, results are presented in Table 2 and compared to untreated CF3 ash assays. The total lead content

Table 1. CF3 Continuous Extraction of Pb, Cd, and Cr (mg/l)
Sampled Prior to Electrochemical Metal Removal

Metal	Concentration in Solution	Time
Pb	43.6	0
	69.2	30 min
	54.8	45 min
	158.8	60 min
	96.2	65 min
Cd	2.	0
	2.	30 min
	2.	45 min
	16.	60 min
	10.	65 min
Cr	< 1.	0
	< 1.	30 min
	< 1.	45 min
	< 1.	60 min
	< 1.	65 min

Table 2. CF3 Total Metal Assay (ug/g Ash)

Metal	Before Extraction (4)	After Extraction
Cd	200	20
Cr	420	470
Cu	520	610
Ni	80	110
Pb	5580	3070

decreased from 5580 ug/g ash to 3070 ug/g ash, resulting in a 45% removal of lead from the ash matrix. These results exhibited a 90% decrease for cadmium. The other metals species were not affected by the extraction process; variances observed in the data were due to the heterogeneous nature of the ash solids.

Electrochemical Process

Batch lead plating experiments were carried out with the pilot plant electrochemical cell and utilized a waste extract solution spiked with a lead standard to a known concentration. The waste extract was obtained through extractions using a fly ash obtained from another incinerator, Massachusetts fly ash (MF). The MF extract was spiked with lead standard solutions to a concentration of 243.0 mg Pb/l. The cell voltage was set at 0.6 V, and after five minutes the concentration decreased to 197.0 mg Pb/l, indicating that lead plating is achievable. Further experimentation will investigate the continuous plating process. Higher lead reductions are expected due to an increased plating time (the cell residence time is 12 minutes) and increased lead concentrations in the waste extract due to the recycle stream.

CONCLUSIONS

The residual ashes from MSW incineration contain high levels of hazardous metals, and leachable concentrations of these materials often surpass the EP Tox test limits as mandated by the USEPA. The classification of these materials may be considered hazardous, warranting costly landfilling. This led to the investigation of the extraction and recovery of Pb, Cd, Cr from the residual ashes. A series of kinetics experiments were carried out, developing the characteristic extraction trends of the various ashes. The extraction kinetics of CF3 ash exhibited a peak in the lead concentration curve at an unsteady state time period of 10 minutes, indicating that the ash/extractant reaction should be controlled at these conditions in order to remove a maximum quantity of lead from the ash particle. Electrochemical experimentation exhibited the potential for metals recovery in a relatively pure form through plating techniques. Based on the results of these investigations a pilot plant was designed and constructed. Initial experimentation indicated that the extraction process was capable of controlling the extraction reaction at the optimum setpoint. Total metals analyses performed on the treated and untreated ashes reveal a 45% reduction for lead and a 90% reduction for cadmium. The electrochemical cell plated lead out of solution at a voltage of 0.6 V, and higher lead reductions are expected due to an increased plating time (the cell residence time is 12 minutes) and increased lead concentrations in the waste extract due to the recycle stream. Future experimentation will include the treatment of various other ash residues as well as process optimization to maximize lead and cadmium recovery, including extract recycle.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefor the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred. This work was funded in part by the New Jersey Hazardous Substance Management Research Center, Project INCIN-13.

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SEMI-INDUSTRIAL TESTING OF THE 3R-PROCESS
(HEAVY METAL EXTRACTION, PRODUCT QUALITY)

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ABSTRACT

The 3R process is a method of inertizing flue dust from refuse incineration. Plants equipped with wet flue gas cleaning systems offer particular advantage in this respect. In the first process step, mobile heavy metals are extracted from the dusts by treatment with the acidic solution of the first scrubber. In a second step, the remaining solid residues are recycled into the incinerator to destroy attached organic pollutants.

A pilot plant has been built for testing the process on a semi-industrial scale. It processed 10 % of the filter dusts and the flue gas scrubbing water of a municipal solid waste incinerator having a throughput of 20 t/h.

The following results were achieved in three measurements campaigns of two weeks each:

- The extraction yields of the most important heavy metals, i.e., cadmium, zinc, lead, and copper are nearly constant in the range of liquid-solid ratios and pH levels expected to occur during operation on a technical scale. The attainable extraction yields are about 90 % for cadmium, between 60 % and 70 % for zinc, and around 20 % for lead and copper.

- Compaction and subsequent thermal treatment of the leached 3R residues in the incinerator of a municipal solid waste incineration plant generates products which easy comply with the limits of the Swiss eluate test.

INTRODUCTION

The continuous reduction of emissions from municipal solid waste incineration plants leads to the concentration of pollutants in the residues of flue gas purification step. Their safe final storage thus becomes a problem Special attention must be given to filter dusts (including boiler ashes) because of their high quantities and because of their high contents of heavy metals, such as cadmium and zinc, and organic pollutants, such as polychlorinated dibenzodioxins and dibenzofuranes.

In refuse incinerators employing wet flue gas scrubbing, the hydrochloric acid generated in the first scrubbing stage can be used to extract mobile heavy metals, especially cadmium, from the filter dusts so that the residues exhibit an elution behavior comparable to that of incineration slags, with which they can be later combined.

This is the treatment concept underlying the 3R process (3R being the German acronym for flue gas purification including residue treatment) developed at the Laboratory for Isotope Technology of the Karlsruhe Nuclear Research Center (1). The basic principles of the process and the semi-industrial testing of the most important step in this treatment - heavy metal extraction - are reported below. The quality of the residues obtained is discussed. The second step in the 3R process, thermal recycling, is described elsewhere (2).

THE 3R PROCESS

The flowsheet of the 3R process is shown in Figure 1. The filter dusts accumulated in the dust removal system of a municipal solid waste incinerator are combined in an extraction step with the acidic flue gas scrubbing water from the first scrubbing stage. The flue gas scrubbing water must, however, first be cleaned by ion-exchange of the mercury which otherwise would be adsorbed onto the filter dusts in the extraction process (3,4). Extraction is followed by liquid-solid separation. The scrubbed residual solid is returned into the incinerator and mixed with the grate ashes, mainly to destroy any adhering organic pollutants.

The 3R process serves to remove, as quantitatively as possible, all toxic metals which can be mobilized from the filter dusts. This does not necessarily mean that the highest possible extraction rates must be attained for these metals because ecotoxic metals, which may be present as highly annealed oxides or immobilized in a silicate matrix, can no longer be made available to the ecosphere through normal environmental influences.

The decisive factor determining the effectiveness of the 3R process is therefore only the stability of the 3R products in the environment. A standardized test is used for its assessment.

The toxic heavy metals present in the filtrate are to be removed from this process depending on whether the metals are to be reused or safely

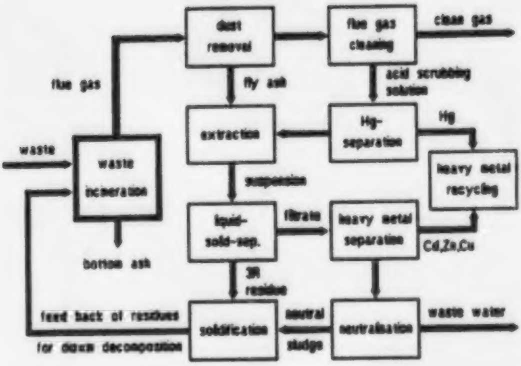

dust removal
flue gas cleaning
clean gas
flue gas
fly ash
acid scrubbing solution
extraction
Hg-separation
Hg
waste
waste incineration
heavy metal recycling
suspension
liquid-solid-sep.
filtrate
heavy metal separation
Cd,Zn,Cu
bottom ash
3R residue
feed back of residues
for dioxin decomposition
solidification
neutral
sludge
neutralisation
waste water

Figure 1. Flowsheet of the 3R process

deposited in a concentrated form. In the former case, ion exchangers can be used, while alkaline precipitation is the most suitable technique to be employed in the latter case. This produces a highly toxic residue of approx. 2 to 4 kg/t of refuse.

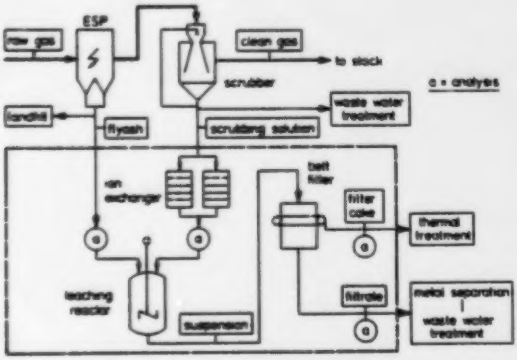
LABORATORY RESULTS

For the first tests conducted on a laboratory scale, original flue gas scrubbing water with a pH of 1 and several electrofilter dusts from incinerators were used. With a liquid-solid ratio of 25 and a temperature of 60 °C, a final pH between 3 and 4 was established in the extraction vessel after some 20 minutes. The average extraction yields obtained were around 90 % for cadmium and 60 % for zinc. Some 30-40 % of the total solid were dissolved (5).

DORA PILOT PLANT

In order to test the process under real conditions, the DORA (DORA being the German acronym for a demonstration plant for residues treatment at Oberhausen) semi-industrial test facility was built at he Oberhausen municipal solid waste incineration plant by Deutsche Babcock Anlagen AG Krefeld (DBA) and the Karlsruhe Nuclear Research Center (KfK). DBA was responsible for constructing and operating the plant, while KfK runned the measurement programs.

The pilot plant was operated in a bypass of the flue gas purification system. Its capacity corresponded to a waste throughput of 2 t/h. Figure 2 shows the flowsheet of the plant. The filter dust and the acidic flue gas scrubbing water were taken directly from the respective parts of the incineration plant. Before entering the extraction reactor, the flue gas scrubbing water were passed through an ion exchange column with an anion exchanger specific to mercury. The extraction reactor is made up of a cascade of two mixed vessels. A vacuum belt filter is used to separate the liquids from the solids.


ESP
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suspension
filtrate
metal separation
waste water treatment

Figure 2. Flowsheet of the DORA pilot plant

The DORA plant is equipped with automatic measurement and control systems for conveying the filter dust (bulk flowmeter), the flue gas scrubbing water feed (flowmeter, pH-meter), and level control in the mixed vessels cascade. This allows the plant to be operated automatically. In addition, the temperatures and the pH levels are measured in all the other vessels. All measured data were recorded by a data logger for archive purposes.

The filter dust feed system contains an automatic sampler which allows representative sampling of the feed flow to be carried out at a partition ratio of 1:1000. All other flows must be sampled manually. Intervals between samplings were around 30 minutes. The metals contents of the solid samples were determined by means of a high-speed X-ray fluorescence analyzer on the plant site (6). In addition, the mercury contents of the flue gas scrubbing water was determined continuously by means of an in house designed analyzer. These measurement are not be described in detail in this paper.

The DORA pilot plant is installed in two containers, one of which accommodates the ion exchange columns and extraction vessels, while the other contains the vacuum belt filter. The data acquisition system and the analytical installations are housed in a third container.

RUNNING THE EXTRACTION EXPERIMENTS

Three measurement campaigns of two weeks each were carried out with the DORA pilot plant during 1987 and 1988. The operating data and their ranges of variation are listed in Table 1. The tests were conducted in order to determine the dependence of the extraction process on the liquid-solid ratio and on the pH levels in the flue gas scrubbing water and in the second extraction vessel within fluctuations normally occurring in plant operation.

TABLE 1. OPERATING DATA OF THE DORA PLANT

Flue gas scrubbing water	700 l/h
Filter dust	40-90 kg/h
pH, flue gas scrubbing water	0.3-1.2
pH, extraction vessel	1.2-4.5
Retention time	1 h
Temperature	50-63 °C
Scrubbing solution (pH 7 or 3)	150 l/h

The flue gas scrubbing water available at the Oberhausen municipal solid waste incineration plant generally had a pH between 0.3 and 0.5. This high acidity resulted in undesirably large amounts of dissolved solids. An initial pH of 1 was therefore set by the addition of NaOH. This was done during the last test campaign.

The 3R residues were washed on the vacuum belt filter, generally with 150 l/h of water. A series of measurements were conducted to find out whether scrubbing with a weak acidic solution (pH 3) had a favorable effect on the quality of the extraction.

On test days, the DORA plant was first operated at steady-state for about 2 hours; the following sampling time was between 6 and 8 hours.

The extraction yields of the different metals were calculated on the basis of the contents of these metals in the filter dusts and in the 3R residue, the fraction of dissolved solid being taken into account. In most cases, the latter was determined in the laboratory with original substances as a function of the pH and the liquid-solid ratio. At initial pH levels in excess of 0.7, the dissolution of solids can be calculated from the titanium concentrations in the two solid streams because, under these conditions, the titanium compounds existing in the filter dust practically are not dissolved.

RUNNING THE ELUATE TEST

As outlined above, the effectiveness of the 3R process can be evaluated only by the quality of the residual products. This evaluation is done by an eluate test, which provides information about the mobility of the toxic substances over prolonged periods of time. There is no international agreement on the method furnishing the correct results for such evaluation. We have decided to apply the eluate test proposed by the Swiss Department of the Interior (7), which has won a certain amount of recognition in Europe in the recent past.

The solids to be examined (100-200 g) should be compacted to a body, whose volume-to-surface ratio is about 1 cm ± 20 %. Also pulverized substances can be tested. The medium used is demineralized water at room temperature, which have a low acidity due to the introduction of CO_2. The mass ratio of the medium relative to the sample must be maintained at 10. The test takes 48 hours; the media are replaced after 24 hours. The arithmetic mean of the concentrations of individual pollutants, multiplied by a solubility factor, must remain within defined limits (7).

This test was applied both to the intermediate product of the 3R process, namely the extraction residue (as a powder), and to the end product, the compacted 3R residue finally passed over the incineration grate.

RESULTS OF EXTRACTION MEASUREMENTS

Aside from some initial difficulties (such as pump wear, plugged dust feed lines, fouling or wear of the pH electrodes), the DORA plant was operated without any problems in accordance with the set operating parameters. The very first measuring campaign, which allowed only slight fluctuation in the operating parameters, fully confirmed the results worked out in the laboratory (8,9). The reproducibility of the extraction yields found for the most important elements can be called satisfactory. One single test day will be described below to demonstrate the stability attained in plant operation and the variance of the results.

On the day chosen, a filter dust mass flow of 80 kg/h and a constant pH of 1 of the flue gas scrubbing water were set; this corresponds to a liquid-solid ratio of 8.75. Figure 3 shows that this target was reached sufficiently well and the setting varied by less than ± 10 % during the day of the test. The pH of the flue gas scrubbing water was set to a similarly constant level, as is shown in Figure 4. In the same diagram the pH in the second extraction vessel is correlated with the variations of the liquid-solid ratio.

Figures 5 and 6 indicate the concentration curves of cadmium and zinc in the solid streams. The fluctuations encountered in the feed concentrations can be considered to be typical. Comparison of the two diagrams shows that the feed concentrations of the two metals are correlated. Such correlation generally applies to all metals which can be thermally mobilized in

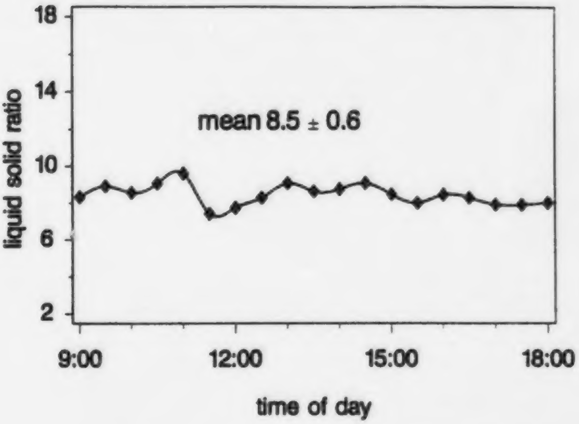

Figure 3. Development of the liquid-solid ratio

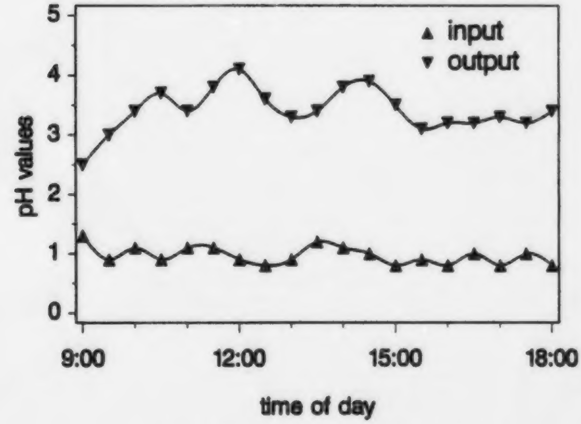

Figure 4. Development of the pH levels

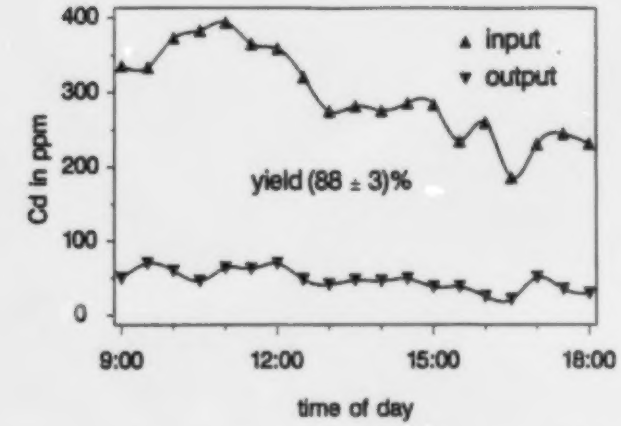

Figure 5. Concentration of cadmium in the solid mass flows

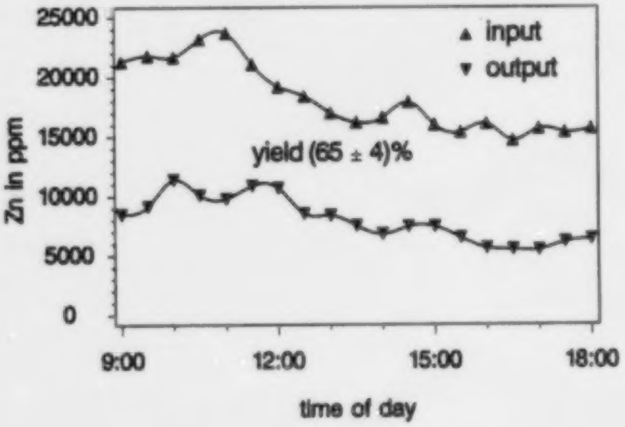

Figure 6. Concentration of zinc in the solid mass flows

refuse incineration, such as copper, lead, or tin, the reason being identical transport mechanisms.

Despite the pronounced fluctuations in the feed concentrations, the reproducibility of the mean extraction yields shown on the diagram is rather good for an industrial process. Similar variances for the two elements were found on almost all test days. For metals of less solubility, such as copper and lead, the error in the mean values increases distinctly, as it is influenced more strongly by errors in sampling and in determining the fraction of dissolved solid.

It had already been recognized in the laboratory experiments that the extractability of the elements depends on the liquid-solid ratio and on the pH established during extraction. This pH, in turn, depends on the liquid-solid ratio, but also on the basicity of the filter dusts. The latter appears to be constant over long periods of time, but may vary considerably in between. On one day, e.g., a material which showed a neutral reaction in contact with water was found for almost two hours. It has not been possible so far to discover the reason for such drastic changes.

Figure 7 combines all extraction yields as of cadmium and zinc metals as a function of the liquid-solid ratio measured in DORA. Practically, a slight uniform rise in the extraction yields of both metals is seen with increasing dilution of the slurry in the mixed vessels.

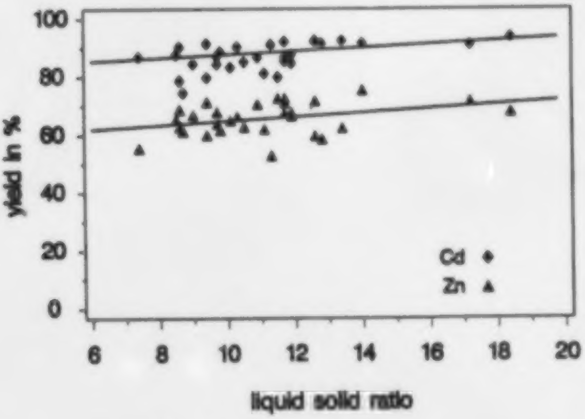

Figure 7. Extraction yields for cadmium and zinc as a function of the liquid-solid ratio

A striking phenomenon is the partial high scattering of the measured data, which has several causes. First of all, reproducibility of the results measured in the first measurement campaign was comparatively low, as can be seen from the error margins of certain values. Secondly, tests have been summarized here which were based on very different initial pH levels, which resulted in different degrees of solid dissolution. A third cause are the different filter dust qualities mentioned above. Throughout the second campaign low heavy metal concentrations were found in the dust. At the same time, the flue gas scrubbing water had pH levels between 0.3 and 0.5, which caused dust dissolutions of 30 % and more. The resulting extraction yields were, e.g., 10 % lower for cadmium.

To study the influence of the final pH on the extraction process, one must obviously start at a constant initial pH. Figure 8 shows the extraction yields for cadmium, zinc, and copper as found in the third campaign when the pH level of the flue gas scrubbing water was stabilized at 1, as a function of the final pH. This clearly indicates that the pH influences metals extraction in different modes.

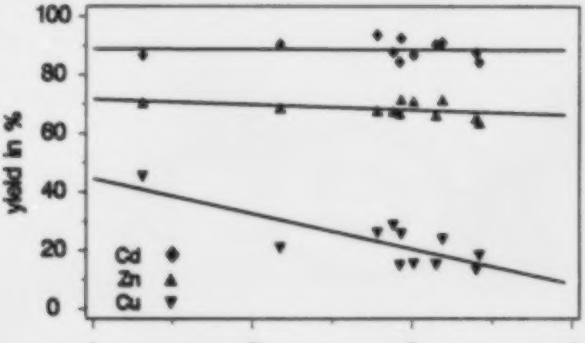

Figure 8. Extraction yields for cadmium, zinc, and copper as a function of the final pH

The extraction of cadmium is practically the same at all pH levels. This is true also of such other elements as potassium, chlorine or bromine, which hardly form sparingly soluble compounds and, consequently, are always easy to leach. The lower the extraction yields, the more sparingly soluble compounds of an element are encountered in the filter dust and the influence of the pH is more pronounced. This is indicated by zinc and copper.

The experimental runs with an acidic washing of the 3R residues on the vacuum belt filter did not produce any significant changes in the extraction results of the metals of interest. However, it can be assumed that such subsequent extraction would improve the elution behavior of the residues, as measured by the eluate tests.

If one summarizes the experience gathered during the extraction experiments with DORA, one must recommend that the process should not be operated with an excessively acidic medium in order to avoid undesirably high dust dissolution. At an initial pH value around 1, approximately constant results are achieved within the framework of liquid to solid ratios of 8 to 15, normally occurring in municipal solid waste incineration plants. This is indicated by the list in Table 2 of the mean extraction yields found in the third measurement campaign. The standard deviations of the means are on the same order of magnitude as the variances encountered in the course of one test day.

TABLE 2. MEAN EXTRACTION YIELDS AT INITIAL pH LEVELS AROUND 1
AND LIQUID-SOLID RATIOS BETWEEN 8 AND 15

Substance	Extraction yield in %
Cd	89 ± 3
Zn	68 ± 4
Cu	18 ± 5
Pb	22 ± 6
Dust	22 ± 5

RESULTS OF THE ELUATE TEST

For a assessment of the effectiveness of the 3R process, the following materials were subjected to the Swiss eluate test referred to above:

- untreated filter dust from the Oberhausen municipal solid waste incineration plant (sample D08),
- 3R residue from the 3rd campaign, washed with water (sample D18),
- 3R residue from the 3rd campaign, washed with diluted acid (pH 3) (sample D28),
- compacted 3R residue, recycled into the incineration chamber of the Oberhausen municipal solid waste incineration plant during a large-scale experiment (2) and removed from the slag discharge belt after having passed the grate (sample D38).

The results of this eluate test are listed in Table 3 for cadmium, zinc, lead, and copper. The data indicate the concentrations in mg/l for the first 24 hours, for the second 24 hours, and the test level, which is the arithmetic mean of the two values. The last column lists the limits applicable in Switzerland.

The test levels obtained from the untreated filter dusts (1st column) for cadmium and zinc are more than one order of magnitude above the limits. In the case of lead and copper metals, this factor is only 3 to 4. The non-compacted 3R residues (columns 2 and 3) indicate a clearly better test level for cadmium which, however, is still a factor of 4 too high. For zinc and lead, levels are found in the region of the limit. The test level for copper of which some 20 % are removed in the extraction process, did not show any modification. This is probably due to the high complexing capability of copper.

TABLE 3. CONCENTRATIONS IN THE ELUATE AND TEST LEVELS OF SOME METALS IN ACCORDANCE WITH THE SWISS ELUATE TEST (DATA IN mg/l)

		D08	D18	D28	D38	limit
Cd	24 h	7.5	0.6	0.5	0.02	
	48 h	0.7	0.2	0.2	0.01	
	Test level	4.5	0.4	0.4	0.015	0.1
Zn	24 h	210	10.0	5.5	0.9	
	48 h	30	4.5	3.1	0.5	
	Test level	133	7.8	4.5	0.7	5.0
Pb	24 h	6.3	1.0	1.9	0.1	
	48 h	1.4	0.8	1.0	0.1	
	Test level	4.2	1.0	1.9	0.1	1.0
Cu	24 h	2.5	2.6	2.4	0.1	
	48 h	0.5	1.1	1.0	0.05	
	Test level	1.7	1.9	1.8	0.08	0.5

Only thermal treatment in the incineration chamber of the municipal solid waste incineration plant, which is required also to destroy organic pollutants, gives the 3R products the quality needed to pass the Swiss eluate test by almost one order of magnitude.

SUMMARY

The DORA pilot plant was built for semi-industrial testing of the extraction step of the 3R process for inertizing filter dusts from municipal solid waste incineration plants. In three measuring campaigns of two weeks each, the following results were achieved:

- The DORA plant permits stable long-term operation in which the set operating parameters are met satisfactorily.
- The extraction yields of the most important heavy metals, such as cadmium, zinc, lead, and copper, are approximately constant in the range of the liquid-solid ratios and pH levels to be expected in operation on an

industrial scale. The attainable extraction yields are around 90 % for cadmium, between 60 and 70 % for zinc, and around 20 % for lead and copper.

- In order to avoid undesired high dust dissolution, the flue gas scrubbing water should be preconditioned to a pH around 1. In that case, the percentage of dissolved dust will generally be between 20 and 25 %.

- Compaction and subsequent thermal treatment of the 3R residues in the incineration chamber of a municipal solid waste incineration plant following the extraction step result of products which easily meet the limits in the Swiss eluate test.

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EVALUATION OF LEACHATE PROPERTIES AND ASSESSMENT OF HEAVY METAL IMMOBILIZATION FROM CEMENT AND LIME AMENDED INCINERATOR RESIDUES

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ABSTRACT

The ability of type II portland cement and hydrated lime to solidify and stabilize municipal solid waste (MSW) incinerator residues was evaluated. Dynamic column leach tests were performed on the unamended combined residue (92% bottom to 8% fly ash), as well as crushed cement and lime amended residues blended in a 1:5 and 1:19 (dry wt) ratio, respectively. The columns were run in a downflow, fully saturated configuration using a pH 4.0 synthetic acid rain leaching media. Leachate withdrawal was controlled at 0.13 ml/min until final liquid/solid ratios of 0.6, 0.4, and 0.9 was achieved in the cement, lime, and residue control columns, respectively. The leachates were analyzed for heavy metal (Al, Cd, Cu, Pb, Fe, Ni, Zn, Mn, Na, Ca) and anion (SO_4, Cl) concentrations, pH, conductivity, alkalinity, and redox potential. Successful immobilization of all the tested metals except aluminum, lead, and nickel were seen in the cement amended leachates, while only zinc and manganese were leached from the lime amended columns in concentrations exceeding those found in the residue control. Compressive strength tests were performed on three different cement amended matrices (11%, 17%, and 33%) and two lime amended matrices (5% and 10%) as a measure of residue solidification. Average 28 day compressive strengths ranged from 200 to 700 psi in the cement matrices, while the lime matrices achived compressive strengths between 4 and 25 psi. This research represents an ongoing effort in evaluating cement, lime, and asphalt as solidification and stabilization amendments for MSW incinerator residues. The primary focus of these efforts is to minimize metal leachability by optimizing MSW incinerator residue/amendment mix designs, and to evaluate the resulting matrices' potential for reuse applications in the construction industries.

INTRODUCTION

Direct landfilling of municipal solid wastes (MSW) represents society's traditional approach to MSW management. Favorable economics, regulatory policies, and sheer convenience has been the impetus behind such handling and disposal practices (Wiles, 1987) . Past scientific studies have linked numerous incidences of adverse human health to the continued pursuit and operation of traditional MSW landfills (Ehrig, 1983; Jackson & Lynch, 1982). Recent legislation has provided incentives and controls to stem the use of landfilling as the sole means of MSW management while encouraging treatment processes that reduce MSW volume prior to land disposal. Recycling and incineration of the raw MSW represent two viable volume reduction processes. The latter has proved effective in reducing the volume and weight of the MSW by 99 and 70 %, respectively, (Hjelmar, 1987).

Managing the resulting solids from the incineration of MSW is an area of growing public and scientific concern. Much of the emphasis has been focused on solidifying and/or stabilizing the incinerator residues, which are more concentrated with potentially toxic constituents than their raw MSW predecessor. Some available solidification/stabilization processes include the use of thermoplastic, encapsulation, glassification, cement-based and lime-based fixation agents (USEPA, 1979). This study evaluates the solidification/ stabilization of a combined (bottom & fly ash) MSW incinerator residue using only cement and lime amendments. Both amendedments have the potential to physically solidifiy the ash and chemically stabilize the metals as insoluble mineral phases inside the stabilized waste form.

The ability of cement and lime to impede the release of heavy metals from their respective residue matrices depends largely on the physical and chemical properties of the binding mechanism, as well as the matrices' environmental conditions . If the amended waste does not contain sufficient pozzolans for lime solidification, or if organics in the cement residue mixes are excessive, a low strength material will result due to the inability to form complex calcium and aluminum silicates within the matrices. The weaker the matrix, the greater susceptibility to environmental attack and hence heavy metal release (Cote, et al, 1986).

Van der Sloot et al. (1987) identifies the primary release mechanisms of heavy metals from solidified/stabilized residues as outward diffusion from the pore water, and to a lesser extent, dissolution of metals from the matrix surfaces. Important factors affecting those release mechanisms include the stabilized material's alkalinity, open porosity, and surface to volume ratio (van der Sloot et al. 1987; Bishop et al; 1985). In a landfill scenario, these properties are largely affected by the conditions and characteristics of the solidified matrix's chemical environment and hydraulic regime. Therefore, selection of a solidification/ stabilization process for MSW incinerator residues must consider the resulting matrix's chemical and physical properties and the site specific environmental conditions.

The objective of this research was to compare various mixes of cement, lime, and asphalt amended incinerator residues with regard to compressive strength and heavy metal leachability. The former was determined using conventional cylinder tests, while the assessment of metal leachability was based on both dynamic (column) and static (batch) leaching experiments. This paper is an interim report addressing only the results of the strength and column leach tests for the cement and lime amended residues.

Solidification of the combined residue with lime and cement produced a material with a compressive strength commensurate to the percent amendment. Column leaching studies of the crushed materials suggest that lime and cement will attenuate certain metal constituents of the incinerator residue, while promoting the mobilization of other metals under mild acidic leaching conditions.

METHODS & MATERIALS

Amendment Process

The combined incinerator residue used in this study consisted of bottom ash and fly ash generated through the combustion of municipal solid waste by the Lamprey Regional Solid Waste Cooperative Incinerator in Durham, New Hampshire. The mass burn facility (100 tons/day) is comprised of three Consumat® modular starved air incineration units and two low pressure steam generating boilers. Incineration of MSW occurs in the lower starved air chamber at 1400 to 1600°F. Hot gases from incineration are reignited in the upper chamber at 1800°F upon addition of excess air. The Lamprey incinerator serves numerous municipalities which vary in population. Typical characteristics of the raw waste stream are summarized in Table A (Beck, 1988).

Table A

Typical Composition of MSW at the
Lamprey Regional Solid Waste Cooperative Incinerator

Waste Form	Composition (% by weight)
Paper	
Newsprint	9
Corrugated	10
Other	17
Plastic	6
Yard Waste	15
Food Waste	15
Metal	
Non-Ferrous	2
Ferrous	6
Glass	12
Miscellaneous	8

The bottom and fly ashes used in the study were generated by the Lamprey facility over a three months period prior to collection. These residues typify normal operation of the facility. The fly ash was collected from ductwork inside the facility, while the bottom ash was obtained from a temporary storage pile. Consequently, the bottom ash was exposed to a leaching environment, both in the incinerator quench tank and the pile, prior to its use in this study. Both ashes were sieved to remove uncombusted material greater than one-half inch, and mixed into a homogeneous blend of 92% bottom to 8% fly ash by volume. A sieve analysis of the resulting combined residue was performed with the results shown in Figure 1.

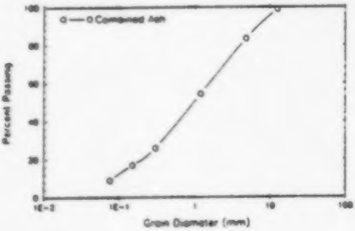


Figure 1: Grain Size Analysis of the Combined Residue

The combined residue was amended with varying proportions of type II portland cement and hydrated lime in a commercially available 6 ft³ barrel mixer. Amended residue mix ratios and corresponding water/amendment ratios (w/a) are given in Table B. The selected mix ratios for the cement amended residues were based on typical cement additions in concrete mixes of a high, medium, and low strength. Similarly, the lime percentages were chosen to mimic the levels associated with incinerator using acid gas scrubbers.

Table B

Incinerator Residue Amendment Composition

Cement Mix Ratio	Lime Mix Ratio	% Amendment (dry wt)	w/a Ratio
1:8	-	11	2.41
1:5	-	17	1.53
1:2	-	33	0.74
1:19	5	-	6.0
1:9	10	-	3.0

Four inch diameter by 8" high cylinders of each sample matrix were created and cured for a 28 days in a fog room. The cylinders were tested for compressive strength at varying time intervals according to standard specifications. The 17% cement and 5% lime matrices used in the column study were fabricated in 42"(w)x 64"(l)x3.5"(d) forms, and cured as previously described. After curing, the cement matrix was crushed to less than one inch using a 6" jaw crusher and sieved to remove particles greater than one-half inch. The 5% lime matrix was crushed by hand, and sieved. The resulting materials were analyzed for size gradation by conventional methods. Figure 2 illustrates the particle size distribution of the crushed amended residues used in the column study.

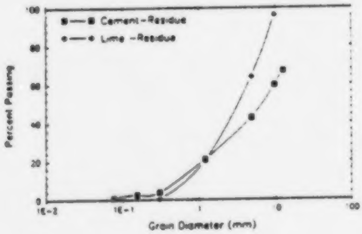


Figure 2: Grain Size Analysis of the Crushed Amended Residues

Column Leaching

Continuous leaching of the crushed amended residues was accomplished in 60"(h) x 10"(dia) schedule 40 PVC columns run in a fully saturated, downflow configuration. Sampling ports were drilled into the column at 12 inch intervals and also served as a means to displace entrapped air during saturation with the leaching media. Three and one-half inch washed sand and gravel filter packs were placed at the bottom of each column separated by a 1 mm mesh. The 17% cement amended residue, 5% lime amended residue, and residue control columns were prepared in duplicate. The control columns contained 26.7 kg dry weight of combined residue, while the cement and lime amended columns contained 36.1 kg and 38.4 kg of combined residue in their respective amendment form. Typical cut column schematics are shown in Figure 3.

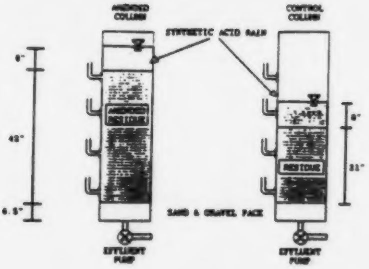


Figure 3: Amended & Control Column Designs

The average packing density for the amended and control columns was 64 and 70 lbs/ft^3, respectively. A pH 4.0 ± 0.3 synthetic acid rain was used as the leaching media. The acid rain consisted of 0.5 mls of 1N H_2SO_4 and 0.125 mls of 0.157N HNO_3 per liter of type II water. The leaching media was allowed to saturate the columns slowly until an eight inch head above the waste forms was achieved. Pumps controlled the column effluent flows at 0.13 ml/min, while the acid rain influents were maintained manually. The effluent flow rate corresponds to an infiltration rate of 53 inches per year. Continuous leaching for the cement and lime amended columns was conducted to final liquid/solid ratios (L/S) of 0.6 and 0.4, respectively, while the control columns achieved L/S of 0.9.

All analyses performed on the column leachate was conducted in accordance with Standard Methods (1985) and ASTM (1988) procedures. Analyses on discrete leachate samples taken throughout the duration of the study included: alkalinity, pH, oxidation-reduction potential (redox), and conductivity. Alkalinity was determined potentiometrically (to pH=4.5) using 1.6N H_2SO_4, 0.16N H_2SO_4, and 0.1N H_2SO_4 titrants for the cement amended, lime amended, and control leachates, respectively. Alkalinity and pH were determined using a Beckman-Altex® meter with a Beckman® Gel-Filled Combination Electrode. Similarly, an Orion MD SA250 Meter with an Orion Redox Electrode was used to determine the leachate's redox potential. Conductivity was measured using a YSI Temperature Compensating Meter.

Metal and anion concentrations were determined from composite samples and reported as discrete analyte concentrations for the given composite period's highest liquid/solid ratio (L/S). Metals analyses were conducted by a commercial analytical laboratory using inductively coupled argon plasma emission spectrometry (ICP). Concentrations of calcium, aluminum, zinc, cadmium, copper, nickel, manganese, lead, and sodium were determined from composites samples filtered through a pre-rinsed (type II water)

0.22 um membrane and acidified using concentrated reagent grade HNO_3. All glassware used in metal sample preparation was soaked for a minimum of four hours in 50% HNO_3, and rinsed 7 times with deionized water followed by two type II water rinses. Chloride, bromide, nitrate, phosphate, and sulfate concentrations were measured on a Dionex® Model 4000i Ion Chromatograph. The anion samples were also filtered through a pre-risned 0.22 um membrane and an additional cation exchange resin prior to injection to minimize fouling of the instrument's separator column.

QA/QC Protocol

Replicate measurements of alkalinity, pH, redox, conductivity, metal and anion concentrations were accomplished randomly on various leachate samples. The relative standard deviation (%RSD) between replicates of each analyses fell within ± 7.2%. According to analytical protocol, known concentration standards for each metal and anion were run after every sixth sample to evaluate instrument accuracy. Additionally, several spiked blank and leachate samples were randomly submitted for metals analysis as a performance check of the commercial laboratory. The results of performance audits and the instrument accuracy checks are given in Figure 4 as average percent recovery.

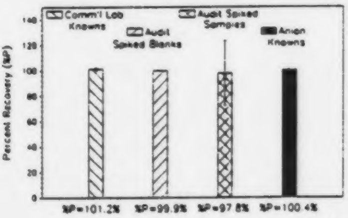


Figure 4: Accuracy of Metal and Anion Analyses

The detection limits of ICP and ion chromotography (IC) were based on the results obtained from the analysis of known standards. This ignores the possible effects of matrix interference, which may be significant given the diverse composition of residue leachate. Table C outlines the detection limits used in this study for each analyte, as well as the accepted detection limits for ICP by the US Environmental Protection

Agency (USEPA, 1982).

Table C

ICP Detection Limits (DL)

Element	Experimental DL (ug/l)	USEPA Estimated DL (ug/l)
Al	52	45
Cd	9	4
Ca	100	10
Cu	20	6
Fe	10	7
Pb	66	42
Mn	2	2
Ni	8	15
Na	1000	29
Zn	8	2

RESULTS & DISCUSSION

Compressive Strength

Figure 5a & 5b contain the results of the compressive strength tests conducted on the various cement and lime amended specimens.

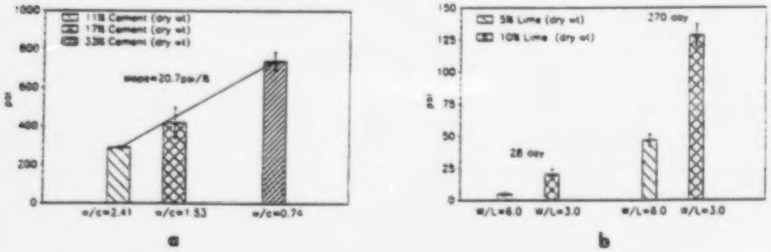


Figure 5: Compressive Strength Results
(a) Cement Amended Specimens - 28 day (b) Lime Amended Specimens - 28 & 270 day

As expected, 28 day compressive strength increased with increasing cement and lime content. Of particular interest was the linear relationship between compressive strength and cement content depicted by the sloped line in Figure 5a. Every 1% increase in cement amendment corresponded to an increase in compressive strength of approximately 20 psi within the given percent range. As illustrated in Figure 5b, a similar increasing strength relationship existed between lime amended specimens cured for 28 and 270 days. The 5% and 10% lime matrices each increased strength at an average rate of 0.34 and 0.06 psi/day, respectively. This slow, steady increase in strength over time is characteristic of a pozzolonic reaction involving lime.

The relatively low range of compressive strengths for the cement amended specimens was expected due to the lack of conventional course aggregate and the possible presence of organic matter in the matrix. These amended mixes were designed to meet desired solidification/ stabilization requirements and were not intended to yield high strength matrices. Past studies reporting higher compressive strengths for concrete/ash specimens merely use incinerator residues as a replacement for a portion of the fine aggregate in a conventional concrete mix (Cook, 1983; van der Sloot, 1987). In this study, the only aggregate in both the cement and lime amended specimens was the combined residue itself.

The compressive strengths of the lime amended specimens were also expected to be low due to the relatively weak pozzolonic activity of the residue, compounded with the small lime percentage in the amended matrix. According to Cullinane & Jones (1989), 20 - 30 % lime content is required to produce a strong pozzolan. The chosen lime concentrations for each matrix were based on typical lime contents seen in the residues produced from incinerators using flue gas cleaning materials (Forrester, 1988).

Environmental Parameters

It is important to note that the crushing of the solidified amended residues, and subsequent column leaching, represents a worse-case leaching senario. Leaching kinetics of monolith-like waste forms would result in much lower concentrations of constituents in the leachate. Further, no attempts have been made to correct for the slight increased mass of combined residue in the amended columns. This difference provides a even a more conservative estimate metal leachability from the amended residues relative to the control residues.

The environmental conditions surrounding any waste material have a profound effect on leachability. Several researchers have attributed metal leachability to the waste forms' physical and chemical properties as well as the characteristics of the hydraulic regime surrounding the waste (Bridle et al. 1987; Cote, 1986). Some of the environmental characteristics of importance in predicting leaching potential of a waste form are alkalinity, pH, redox, and conductivity (Constable et al, 1984). These parameters were measured throughout the column study and are reported in Figures 6a-e as a function of liquid/solid ratio (L/S).

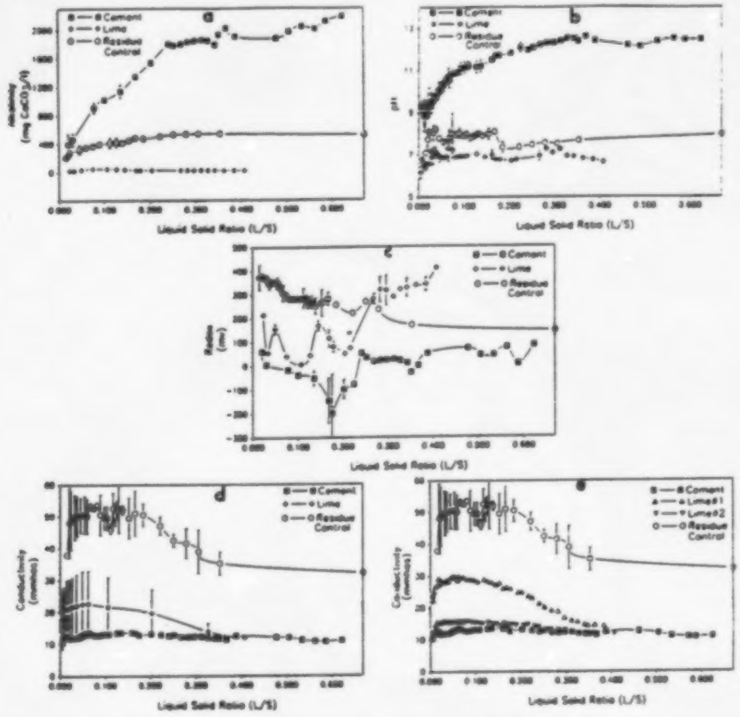


Figure 6: Environmental Parameters Measured on Column Leachates vs L/S
(a) Alkalinity (b) pH (c) Redox (d) Average Conductivity (e) Conductivity for Each Lime Replicate

Previous studies indicate that the leaching of heavy metals from a solidified/stabilized waste form is governed by the amount of alkalinity present in that waste (Bishop et al, 1985). The higher the buffering capacity provided by amending agent, the more resistant the stabilized waste form will be to attack from acidic rain or soil water. However, extremely buffered systems containing caustic alkalinity can cause the release of metals that are amphoteric and soluble at high pH levels. Figure 6a shows the trends in leachate alkalinity from the 100% combined residue (control), 17% cement, and 5% lime columns. The alkalinity of the combined residue and lime amended leachates increased slightly at first, then remained relatively constant throughout the study. An apparent anomaly exists between the alkalinities measured in the residue control and lime amended leachates. The addition of 5% lime to the residue caused a decrease in alkalinity and pH. This may be due to bicarbonates in the unamended residue reacting with the hydroxide alkalinity from the lime addition to form carbonates which readily precipitate out in the amended column as calcite ($CaCO_3$). Conversely, the cement amended leachates demonstrated a sharp increase in alkalinity through a L/S of 0.22, followed by a continual, more gradual increase through the rest of the leaching period. The pH (Figure 6b) of the respective leachates also followed similar trends. This suggests buffering of the acidic leaching media occurs rapidly to pH and alkalinity levels characteristic of the waste form itself.

The pH ranges for the control leachates were similar to those cited in the literature for incinerator bottom ash (Eighmy et al, 1988). This was expected given the composition of the combined ash (i.e. 92% bottom to 8% fly ash). The control leachate alkalinities exceeded published data by a factor of 10 (Hjelmar, 1987; Cundari & Lauria, 1987; Eighmy et al, 1988). The lack of agreement may be due to variations in the waste streams or operating procedures at the various incinerator facilities cited.

Figure 6c illustrates the redox potential measured in each of the leachates. In general, all redox measurements experienced a degree of variability not only with time, but also between replicate columns. This is not uncharacteristic of platinum probe redox measurements, especially in solutions containing such wide ranging constituents (Eighmy et al, 1988). The cement amended leachates were consistently more reduced than the lime amended and control leachates. This, coupled with the high pH and alkalinity exhibited by the cement amended leachate should render most heavy metals immobile and resistant to environmental leaching from the matrix (Cote et al, 1986). More reliable redox data, in concert with pH and alkalinity, should enable better prediction of leachability from specific waste forms and also lend insights into the metal speciations.

Figure 6d & 6e illustrate the measured leachate conductivites for the cement, lime, and control columns. The cement amended leachate demonstrates a relatively low and consistent conductivity compared to the lime amended and control leachates. This may be due to adsorption of the dissolved solids to the cement paste matrix and the propensity of the silicates in the cement to form insoluble complex precipitates that deposit throughout the column (Cook, 1988). As seen by the wide error bars in Figure 6d, the replicate lime columns demonstrated significantly different leachate conductivities. Therefore, the leachate conductivities for each lime amended column were plotted separately in Figure 6e. Initially, channelling in lime column #2 was suspected as the cause of this variability. Although this may have been a factor, the principal source of the variability was probably due to a non-homogeneous mixture of the combined residue with respect to sulfate concentrations (see Figures 8a & b).

Anion & Heavy Metal Leaching

The anions and heavy metals selected for analysis were based on the results of previous studies and existing/proposed drinking water standards established by the USEPA. These constituents do not fully represent the entire spectrum of anions and metals associated with the incinerator residue, but hopefully encompass the principle metal/anion species. As a test, Figure 7 was developed comparing the summation of measured cation equivalent concentration in the leachates (meq/l) to their corresponding anion equivalent concentrations. [Note: anion analysis for the residue control leachate was not completed prior to publishing deadline]

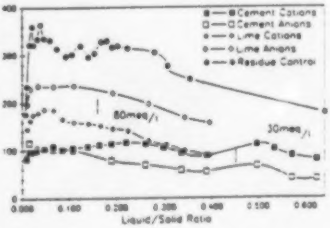
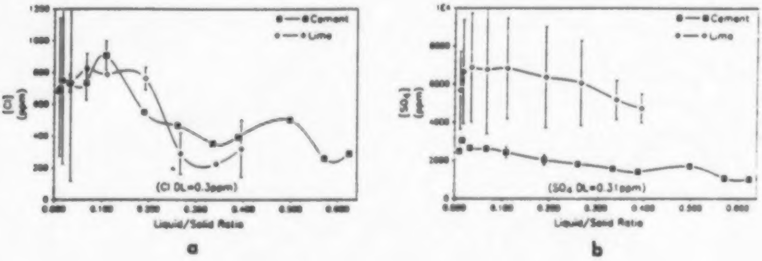


Figure 7: Comparison of Cation and Anion Equivalent Concentrations

A comparison of the cement cation and anion equivalents plots indicate that the principle heavy metals species released through a L/S of 0.1 are in the form of metal chlorides and sulfates. After this L/S, a constant difference of 30 meq/l is apparent throughout the study's duration, thus suggesting that certain anionic species that are leached in significant concentrations from the waste forms are going unmeasured. The relatively alkalinity associated with the cement amended leachate suggests that the unmeasured anions are carbonates, bicarbonates, and hydroxides. Further analysis for these anions, phosphates, and fluorides are warranted (van der Sloot et al. 1987).

A similar comparison of the lime cation and anion equivalents show a different relationship. From the study's beginning, anion concentrations exceeded cation concentrations by approximately 80 meq/l. This trend persisted through the length of the study (L/S=0.4), suggesting that metals, other than those evaluated within this study, may have been present. Further analysis of the lime amended leachate for potassium, magnesium, chromium, molybdenum, and strontium may provide additional insights into cation balance and speciation (Miner et al. 1986; van der Sloot et al, 1987; Hjelmar, 1987)

The results of the anion analyses are given in Figures 8a & 8b. As with conductivity, wide variability between replicate lime amended leachates existed for sulfate, and to a lesser extent, chloride concentrations. Both analytes showed an overall decreasing concentration with time. Replicate cement amended leachates showed chloride variation in the initial stages of the study, however this variability decreased with increasing L/S. Sulfate concentrations for cement amended leachates were repeatable for each replicate column, and demonstrated a slight decreasing trend.



a

b

Figure 8: Average Anion Concentrations Measured in the Amended Column Leachates
(a) Chloride vs L/S (b) Sulfate vs L/S

The concentrations of lead, aluminum, calcium, nickel, zinc, manganese and sodium in the residue control, cement, and lime amended leachates are presented in Figures 9 - 15. Note that all of these metal except for calcium, and manganese are either regulated or facing future regulation under the Safe Water Drinking Act & Amendments (SWDA) by the USEPA (Camp Dresser & McKee, 1987).

Recent leaching studies cite maximum lead concentrations for combined incinerator residue leachates at approximately 0.1 ppm (Eighmy et al, 1988; Hjelmar, 1987). Lead concentrations in this study were measured at similar trace levels. This suggests lead is typically immobile at these common pH and redox values. A dramatic increase in lead for the cement amended leachate (≈ 9-15 ppm) is shown in Figure 9 at a L/S of 0.25.

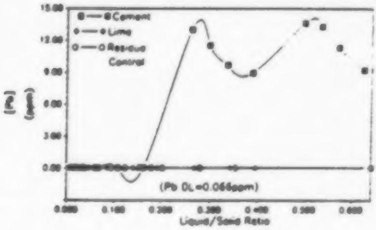


Figure 9: Average Lead Concentrations in the Amended and Unamended Residue Leachates

At this L/S, the average pH and redox were measured to be 11.6, and -100 mv. Lead releases under these environmental conditions could have been associated with the reduction of iron oxides and lead chromates as well as the dissolution of lead solids (e.g. lead hydroxides & lead carbonates) at high pH levels (Theis & Richter, 1979; Kinniburgh et al, 1976). The amphoteric nature of lead at differing pHs suggests that cement stabilized residues may leach lead in environmentally significant concentrations especially when the integrity of the monolith has been compromised.

Aluminum concentrations (Figure 10) in the cement amended leachates varied significantly from those found in the control and lime amended leachates. The relative insolubility of aluminum in the control and lime amended columns was expected since their pH conditions were near the point of minimum hydroxide solubility. Conversely, the cement amended columns operated within a pH range that was favorable to the formation of negative hydroxy-complexes of aluminum, e.g. $Al(OH)^{-}_{4}$ (Stumm & Morgan, 1981). The mechanism of aluminum release from the cement amended matrix certainly requires further analysis, especially considering the recent concern about possible links between aluminum and various neurological disorders (Miller et al, 1984).

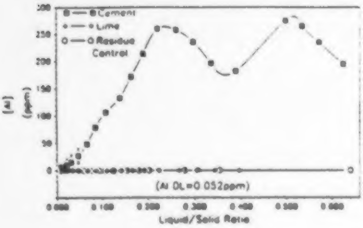


Figure 10: Average Aluminum Concentrations in the Amended and Unamended Residue Leachates

The release of calcium (Figure 11) from the three waste forms appears to mirror the release patterns of aluminum.

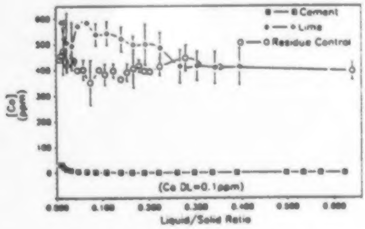


Figure 11: Average Calcium Concentrations in the Amended and Unamended Leachates

Significant calcium concentrations were measured in the control and lime amended leachates, while the cement amended leachate exhibited trace levels of calcium for the majority of the study. This suggests that a possible ion exchange process may be occurring between the aluminum in the residue and the calcium in the lime amendment.

Trace concentrations of nickel (Figure 12) and zinc (Figure

13) were measured for all the column leachates during the study.

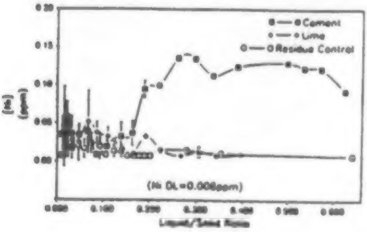


Figure 12: Average Nickel Concentrations in the Amended and Unamended Leachates

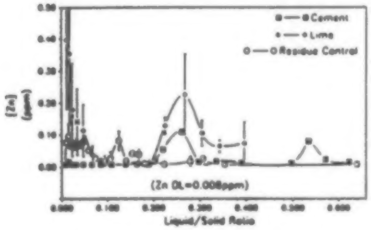


Figure 13: Average Zinc Concentrations in the Amended and Unamended Leachates

The concentrations determined for the combined residue leachates were consistent with levels cited in the literature (Cundari & Lauria, 1987; Hjelmar, 1987). Initial measurements of nickel and zinc in all column leachates were highly variable during initial leaching, but variability appeared to decrease with increasing L/S. Nickel concentration in the cement amended leachate exceeded the control by approximately 0.7 ppm at a L/S greater than 0.2. This may be due to nickel being first adsorbed to iron oxide surfaces and being released under reducing conditions (Theis & Richter, 1979; Feijel et al. 1988).

In Figure 14, greater distinctions between amended matrices and the residue control are seen with respect to manganese.

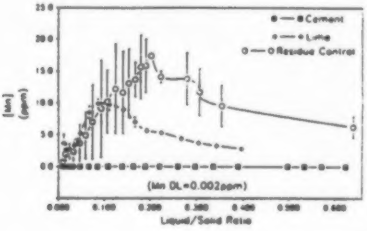


Figure 14: Average Manganese Concentrations in the Amended and Unamended Leachates

Replicate manganese concentrations in the control columns experienced a much greater degree of variability than did the amended leachates. It appears that significant attenuation of manganese was accomplished by the cement amendment, while the lime amendment demonstrated manganese immobilization for L/S greater than 0.18.

Cement and lime amendments experienced similar success in attenuating sodium. Figure 15a illustrates the average sodium concentrations associated with each column leachate. The average sodium concentrations measured in the cement and lime amended leachates were significantly lower than the residue control concentrations, but not significantly different from each other. This was due to the high variability in the lime amended results. As with the conductivity results, separate plots illustrating the sodium concentration respective of each lime amended column are plotted in Figure 15b. The ability to immobilize even small percentages of the total leachable sodium fraction from incinerator related residues may prove significant given the proposed action by the USEPA to regulate its concentration.

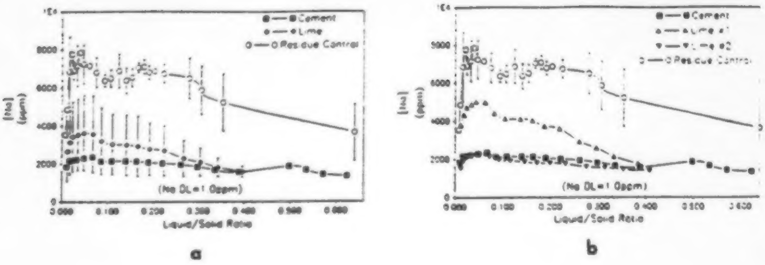


Figure 15: Sodium Concentrations in the Amended and Unamended Leachates
(a) Average [Na] (b) [Na] of Each Lime Replicate

Other metals monitored during the column study included cadmium, copper, and iron. Concentration plots were not developed for these metals since their concentrations in the leachates generally fell below detectable limits.

SUMMARY

This research represents an ongoing effort in evaluating cement, lime, and asphalt as solidification and stabilization amendments for MSW incinerator residues. Cement and lime results to date indicate both amendments show promise with regard to their strength properties and heavy metal immobilization potential given the range of mix ratios and leaching performed. Compresssive strengths for three different cement amended residues ranged between 280 and 800 psi. Lime amended residues achieved lower compressive strengths, but were successful in forming a pozzolonic reaction with the residues. In column leach tests, the cement amended residue (17% dry wt) was successful in immobilizing all tested metals with the exception of aluminum, lead, and to a lesser extent nickel. Similarly, the lime amended residue (5% dry wt) immobilized all test metals except manganese and zinc. More aggressive leaching experiments using various blends of cement, lime and asphalt amendments should better define each material's potential to effectively solidify and stablize MSW incinerator residues.

ACKNOWLEDGEMENTS

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"INTERNORDIC" METHOD CALIBRATION FOR SAMPLING AND ANALYSIS OF DIOXINS AND OTHER CHLORINATED ORGANIC COMPOUNDS

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ABSTRACT

Finland, Norway and Sweden are using different sampling methods, spiking protocols and analytical techniques in the analysis of

chlorinated dioxins (PCDDs) and dibenzofurans (PCDFs) from municipal solid waste (MSW) incineration. The data from all the countries are compared but there has been no validation of the different techniques.

In the beginning of october 1988 five different sampling trains were used parallel in a MSW incinerator at Högdalen, Sweden: three of the sampling trains came from Finland, one from Sweden and one from Norway. The study has validated sampling train, spiking protoocol and analytical method which are recommended and used by different Nordic countries.

INTRODUCTION

Polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) have been identified in all municipal solid waste (MSW) incinerators in the Nordic countries. Finland, Norway and Sweden have their own spiking protocol, sampling method and analytical procedure. Emissions of dioxins and other chlorinated carbones from the three countries have been compared to each other but untill now no validated study of the methods from the three countries have been presented. Hagenmaier et al. (1,2) and Marklund et al. (3) have carried out studies of various sampling and analytical method of flue gas for PCDD and PCDF analysis in MSW incinerations but only one method from these countries was represented. In Finland there has been plans to perform such intercomparison studies (4). In this study we have used five different sampling systems three of the systems (system 1,4 and 5) are based on the recommendations from the Swedish National Environmental Protection Board (5), one (system 2) is a high volume sampler (6) intended to take samples during a short period of time and finally one (system 3) is a simplified easy to handle sampling system and specially optimized for collecting aerosols.

DESCRIPTION OF THE FACILITY

Sampling was made at the municipal waste incinerator "Högdalen" (waste fired boiler P-3) in Stockholm, Sweden. The heat output of boiler P-3 is 40 MW, its burning capacity is 15 metric tons per hour and it has been comissioned 1985. The off gases are cleaned by mixing first with dry lime and filtering afterwards in a baghouse.

The stack height over the ground is 140 m. Off gas samples were taken at 40 m level in the stack. The sampling location and technicalities were ideal for simultaneous sampling. Plant instrumentation were used during the sampling. Carbon monoxide, oxygen concentration in the stack and after the boiler, hydrocloric acid, particulates and water content in the off gases as well as combustion temperature were continuously measured during the sampling. The data during the sampling is presented in Table 1.

TABLE 1. GENERAL DATA IN THE SAMPLING

Sampling period Parameter	1	2	3	4
CO (ppm)	< 40	< 40	< 40	< 40
CO_2 (%)	9.5	9.5	9.5	9.5
O_2 (%)	9.5	9.5	9.5	9.5
HCl (out) mg/m^3n	< 50	< 50	< 50	< 40
Combustion temp. oC	830	830	830	830
Flue gas temp. oC	110	110	110	110
Particulates mg/m^3n	< 1	< 1	< 1	< 1
Water in offgases (%)	17	17	17	16
Gasflow m^3n/h	82000	82000	82000	82000

EXPERIMENTAL

SPIKING PROTOCOL

The samples taken during sample period 1 - 3 were pre-spiked with sample spikes from the laboratory which was going to make the analysis. Samples taken during sample period 4 were all spiked with the Umeå sample spike. As sample system 3 and 4 were analysed at University of Umeå they were spiked with the same spike during sample period 1 - 4. The spiking protocol is presented in Table 2.

SAMPLING TRAINS

In the experiments five different sampling arrangements were used. These sampling trains are described in the following text.

a. System 1 and 4: The SNV Sampling method. Figure 1 (5). The Swedish National Protection Board has recomended a sampling technique composed of a glass fiber filter held at 160°C (+/-15 oC) followed by a condenser wcich takes down the temperature of the flue gas to at least 20°C. The condensate is collected in a glass flask and the dry sample gases are passing an adsorbent cartridge containing more than 5g of Ambrelite XAD-2 per m^3 flue gas sampled. The gas

TABLE 2.   SPIKING PROTOCOL OF SAMPLES

Pre sample spike:	$^{13}C_{12}$	2378 - TCDD	2.5 ng
		2378 - TCDF	2.5 ng
		12378 - PeCDD	2.5 ng
		12378 - PeCDF	2.5 ng
		23478 - PeCDF	2.5 ng
		123789 - HxCDD	2.5 ng
		123789 - HxCDF	7.5 ng
		123478 - HxCDF	2.5 ng
		1234678- HpCDD	2.5 ng
		1234678- HpCDF	2.5 ng
		OCDD	2.5 ng
Clean up spike	$^{13}C_{12}$	1234 - TCDD	2.5 ng
Syringe spike	$^{37}Cl_4$	2378 - TCDD	
	$^{13}C_{12}$	123678 - HxCDD	

flow through this sampling train was 2 - 3 m^3/h in our experiments and the total volume collected was 6-10 m^3. Some minor modifications in the sampling train have been made in Finland. The sampling system is schematically presented Figure 1. After sampling the sampling train was thoroughly rinsed with several aliquots of acetone and toluene.

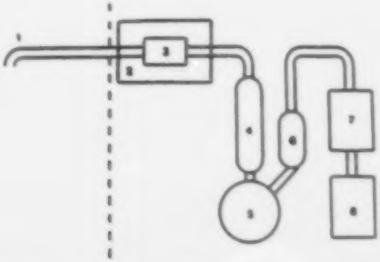

Figure 1. Sampling equipment. System 1 and 4 (SNV recommended) (legend on the next page)

1. Glass tube for isokinetic sampling
2. Oven held at 150 (+/- 15 oC).
3. Glass fibre filter
4. Condensor (below 20 oC)
5. Condensate bottle
6. Adsorbent column (> 5 g XAD-2/m^3 sample)
7. Pump
8. Gas meter

b. System 2: The Finnish high volume sampling method. Figure 2 (6). This system is developed by the Tecnical Research Center of Finnland. The sample is taken isokinetic from the stack with help of a build in null probe. The gas flow passes trough a heated glass line to a heated filter. After the particles have been collected on a glass fiber filter the flue gas sample is cooled down to -30°C (cryostate) in an impinger system followed by an adsorbtion cartridge. During one sample period with the other systems, two samples were taken with the high volume system, one sample with the adsorbtion cartridge filled with Ambrelite XAD-2 and one sample with a poly-urethan foam plug. The flow rate was 5 -10 m^3/h and the total volume sampled was 2-4m^3. The sampling train is schematically presented Figure 2. After sampling the sampling train was thoroughly rinsed with several aliquots of toluene.

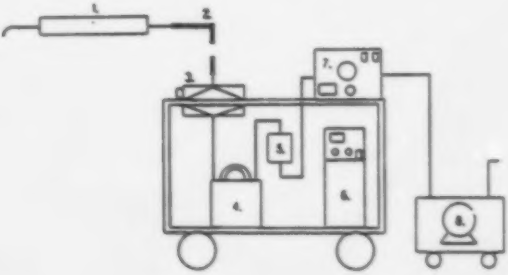

Figure 2. Sampling equipment. System 2, Finnish High Volume Sampling train

1. Glass tube for isokinetic sampling
2. Heated gas line
3. Heated glass fibre filter
4. Condensor (below 20 oC)
5. Adsorbent column
6. Cryostate
7. Controll unit
8. Pump

c. System 3: The University of Umeå sampling method. Figure 3 (3). A water cooled glass probe ensures a cooling of the sample down below 20°C within seconds. In addition to the cooled probe an ice cooled condensate collector are used followded by an impinger filled with toluene. After the impinger the sample gases are passing a poly-urethan foam plug. The flow rate was about 2 m^3/h and the total sample volume 6-10 m^3. The sampling train is schematically presented Figure 3.

After sampling the sampling train was thoroughly rinsed with several aliquots of acetone and toluene.

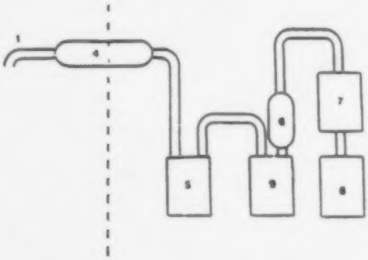

Figure 3. Sampling equipment. System 3, Umeå sampling train

1. Glass tube for isokinetic sampling
4. Condensor (below 20 oC)
5. Condensate bottle
6. Adsorbent column, poly-urethane foam plug
7. Pump
8. Gas meter
9. Impinger system

d. system 5: The Norwegian sampling method (7,8). Particles from the isokinetically sampled flue gases were colleted on a heated glass fiber filter plate. Flue gases were condensed by a double cooler system. More volatile compounds were trapped by an adsorbent column packed with Ambrelite XAD-2. The total sampled volume in our experiments was 13-16 m^3, and the flow rate was $5m^3/h$. The sampling train is schematically presented Figure 4.

After sampling the sampling train was thoroughly rinsed with several aliquots of acetone and toluene.

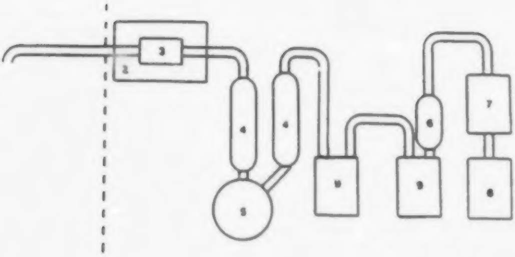

Figure 4. Sampling equipment. System 5, Norwegian standard train

1. Glass tube for isokinetic sampling
2. Oven held at 130 (+/- 5) oC.
3. Glass fibre filter
4. Condensor (below 20 oC)
5. Condensate bottle
6. Adsorbent column (> 5 g XAD-2/m^3 sample)
7. Pump
8. Gas meter
9. Impinger system (successive cooling from 0 oC to -60 oC)

EXTRACTION, CLEAN UP AND ANALYTICAL METHODS

Samples from "sampling system 1" were treated by Domestic Fuel Laboratory of VTT and University of Jyväskylä, Finland. The samples were extracted with toluene and/or hexane, evaporated to dryness and diluted in hexane. The hexane phase was extracted with a water solution of K_2CO_3. The water phase was treated with acetic acid anhydride, extracted with hexane and analysed for chlorophenols on a high resulotion gas-chromatograph (HRGC) equipped with an EC-Detector. The original hexane phase was purified with H_2SO_4 and analysed for Polychlorinated biphenyls (PCBs) and chlorinated benzenes (PCBzs). A small aliquotes was injected on a HPLC and anlayzed for Polyaromatic hydrocarbons (PAHs). The hexane phase was futher purified on an alumina oxid column and a charcoal column and analysed for PCDDs and PCDFs on a HP-MSD quadrupolic masspectrometer (HRGC-LRMS).

Chemical Laboratory of VTT (Helsinki) treated samples from "sampling system 2". The extraction and clean-up methods decribed above was used in this laboratory also but a different analytical technique were preformed. In the analyses of PCBs PCPs and PCBzs a HRGC with a FID Detector was used. For the PAH analyses a HP-MSD, HRGC-LRMS was used.

Samples from sampling system 3 and 4 were taken to Umeå University for analyses. The particles in the condensate and wash fraction were collected on filters and extracted together with filters and adsorbents from the sampling tarains in a soxhlet-extractor with toluene.

The water phases were extracted with methylene chloride and were combined with the wash- and soxhlet extracts. Tetradecane was added as a keeper to prohibit losses of the analyzed compounds. The extracts were evaporated to a small volume and divided into two parts. One part was used after extraction with basic water and further clean up on a silica column for analysis of PCBz on A VG-12 quadrupolic masspectrometer (HRGC-LRMS). The water phase was neutralized and extracted with hexane. The hexane phase was injected on the VG-12 MS system for analysis of PCPs. The second part of the divided extract was purified on a acidsilica - silica column followed by an alumina oxide (Super 1) column an injected on a VG 70 - 250E HRGC-HRMS operating at 5000 dalton in resolution (3).

The samples in "sampling system 5" were taken by Center for Industrial Research, Norway. The filters, condensates and adsorbents were extracted with toluene and second time with acetone.The toluene extracts were divided into two parts; one part was analyzed for PCDD and PCDF by the Norwegin Institute for Air Research using a clean up analyzing method described elswhere (7,8), the other part was added half of the acetone extracts and analyzed for polycyclic aromatic hydrocarbons (PAH) and chlorinated benzenes and phenoles by Center for Industrial Research. The PAH components were analyzed by GC/FID using internal standards for quantification according to Bjoerseth (9). The chlorinated benzenes were analyzed by GC/ECD after a clean up including H_2SO_4- and KOH/ethanol tratment of the extarcts. Prior to the analyse of the chlorinated phenols the condensate were extracted several times with cyclohexane at pH=2. The phenols were derivated with acetic acid anhydride and detected by GC/ECD. Quantification was made by internal and external standards.

Quantification and identification of PCDD/PCDF was carried out by mass spectrometric detection using electron impact and negative ion chemical ionization. Chemical analysis of PAH-compunds started by the extraction of the filter and adsorbent with toluene in a Soxhlet apparatus. The condensate was liquid-liquid extracted with toluene. Afterwards the extracts were fractionated into non-polar and moderately polar fractions by normal phase high performance liquid chromatography (HPLC) on a semipreparative uPorasil column. The compound identification and quantification was made by HRGC-mass spectrometry (8).

RESULTS AND DISCUSSION

Polyaromatic hydrocarbons (PAH) were analyzed by three of the four laboratories. The levels of the PAH components were in practically all cases below the detection limit, which was set to 100 ng/m^3n. Therefore no resonable comparison could be made.

Polychlorinated biphenyls (PCB) components were analyzed by three laboratories as well. The levels found were very close or below the detection limit. Therefore no reasonable comparison could be made.

The analyses of clorinated benzenes show that all analyses give almost identical concentrations in the samples. The difference in results is on the order of one magnitude at its maximum. However, the concentrations are so low (1 - 2 Ig/m^3n) that this difference is to be considered negligible. The results of analyses are presented in Table 3. below.

TABLE 3. COMPARISON OF SAMPLING AND ANALYSIS OF TOTAL CHLORINATED BENZENES (PCBz) ug/m^3_n

Sample period	1	2	3	4	Detection limit/isomer
System 1	1.5	1.6	1.6	1.5	0.002 - 0.02
System 2	0.7; 1,8	NA	NA	1.9	0.6 - 0.9
System 3	1.2	1.0	2.0	1.5	0.001
System 4	0.3	2.4	1.1	2.2	0.001
System 5	1.5	0.5	1.4	0.4	0.002 - 0.005

NA = not analysed

The analyses of polychlorinated phenols show a good agreement to each others. However, the high sum of PCP's in System 4, samples 3 and 4, can be explained on the basis of exeptionally high concentrations of 2346 tetrachlorinated phenols. The detection limit in practically all analysis has been low.

The concentrations of PCP-components have been low in all samples, which also have caused some analytical difficulties. This makes also the comparison of different results uncertain. The results of analysis for polychlorinated phenoles are presented in Table 4. below.

TABLE 4. COMPARISON OF SAMPLING AND ANALYSIS OF TOTAL CHLORINATED PHENOLS (PCP) ug/m^3_n

Sample period	1	2	3	4	Detection limit/isomer
System 1	1.0	0.9	1.1	1.4	0.002 - 0.02
System 2	3.0; 3.1	NA	NA	6.4	0.3 - 0.4
System 3	2.6	1.3	2.2	2.5	0.001
System 4	1.2	0.6	7.1	5.1	0.001
System 5	0.6	0.5	1.3	0.7	0.01 - 0.02

NA = not analyzed

Due to the extreme low concentrations of PCDD/PCDF compounds in the samples only 2378-TeCDF could be found in the Finnish analysis. In Sweden, however, the analysis were performed with a high resolution magnetic mass spectrometer, HRMS. This instrument could detect also all other isomers in the samples in low pg/m^3 level. The instrument used in Norway was also more sensitive than the Finnish.

TABLE 5. COMPARISON OF SAMPLING AND ANALYSIS OF 2378-TeCDF COMPENSATED TO THE SAMPLING RECOVERY. pg/m^3_n

Sample period	1	2	3	4	Detection limit/isomer
System 1	14	7	7	13	1
System 2	NA	NA	NA	NA	
System 3	8	14	10	13	0.1
System 4	7	9	7	25	0.1
System 5	7	6	12	0.7	0.1 - 1.0

NA = not analyzed

In Table 5. the levels of 2378-TeCDF are shown in different samples during different sampling periods. Good agreement can be seen. It must also be pointed out that sampling systems 1 and 4 , which are identical, show a good agreement. System 1 was analyzed in Finland and System 4 in Sweden.

In Table 6. the TCDD equivalents are calculated using three different approaches: "Eadon" (most commonly used in Scandinavia), Nordic (the new basis for calculations in Scandinavia) and the US-EPA method published 1985.

The detection limits are also calculated on TCDD equivalent basis in Table 6.

Table 6 shows that the equivalent concentrations are very low. Even if the differences in results can seem to be large, in only one case the real difference in results is obvious. The lower the PCDD/-PCDF concentrations in the off gases are the more critical is the proper sampling.

TABLE 6. COMPARISON OF SAMPLING AND ANALYSIS OF CHLORINATED DIOXINS AND DIBENZOFURANS EXPRESSED IN TCDD-EQUIVALENTS COMPENSATED TO THE SAMPLING RECOVERY. $ng/m^3{}_n$

Sample period	1	2	3	4	Detection limit
System 1					
nordic model	<0.03	<0.03	<0.03	<0.03	0.03
Eadon	<0.03	<0.03	<0.03	<0.03	0.03
System 3					
nordic model	0.020	0.037	0.034	0.019	0.002
Eadon	0.019	0.029	0.027	0.018	0.002
US-EPA	0.014	0.020	0.019	0.014	0.002
System 4					
nordic model	0.013	0.014	0.012	0.073	0.002
Eadon	0.012	0.014	0.014	0.064	0.002
US-EPA	0.010	0.010	0.010	0.041	0.002
System 5					
nordic model	0.02	0.04	0.03	<0.01	0.01
Eadon	0.02	0.04	0.030	<0.01	0.01

Fluctuations in PCDD/PCDF concentrations due to particulates, not isokinetic sampling and sample contamination (in the field operations and in the laboratory) will increase in significance when the levels of these components are extremely low.

This is probably the reason why in one of the samples, Sample period 4 - Sampling system 4, the concentrations are remarkably higher than in other samples. This indicates that replicate sampling is to be recommended when PCDD/PCDF levels are expected to be very low in samples.

The use of different sampling methods and analytical technology gives comparable results. Table 7 shows the recovery of the "pre-sample spike" added to the sample train before sampling period 4.

The recoveries are high and very comparable to each others. This implicates good sampling and handling of the samples, which is usually the case in intercalibration studies. Results also show that the sampling systems 1, 3 and 4 give comparable results. The comparison of total recovery in sample 4 is presented in Table 7.

TABLE 7. COMPARISON OF TOTAL RECOVERY IN SAMPLE 4.

recovery % $^{13}C_{12}$-lable	system 1	system 3	system 4	system 5
TCDD	81	100	107	
PeCDD	92	86	89	
HxCDD	73	63	84	
HpCDD	22	65	94	
OCDD	20	18	59	
TCDF	90	87	96	
PeCDF	68	71	85	
HxCDF	53	28	45	
HpCDF	31	1.9*	20*	

* not reliable due to instrumental problems

CONCLUSIONS

This "Internordic" method calibration turned out to be very succssfull joint project. The common opinion among the participants is that joint method and sampling "calibrations" should be arranged frequently also with other laboratories.

The systems tested were suitable for sampling polychlorinated carbones and PCDD/PCDF compounds at field conditions. However, some of the used sampling equipments were more "research equipments" than practical field samplers. It is obvious that the sampling systems should be developed towards more "handy", light weight trains. The "sampling system 3" is an example of that line of development.

The analytical procedures used in sample treatment and quantification of compounds were somewhat different. The spiking protocol used differed and in two cases no pre-spiking was used at all. However, the results show that proper sample treatment, good laboratory work and careful quantification gave good agreement between different samples and "sampling systems". As a conclusion of this study it can be said that the results from different Scandinavian works for chlorinated aromatic compounds are comparable.

In the future, new technologies must be developed to measure and analyze with sufficent accuracy the very low concentrations of chlorinated carbones and TCDD/TCDF compounds in the off-gases from municipal solid waste incineration.

The work described in this paper was not funded by the U.S Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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COMPARATIVE STUDY FOR VALIDATED SAMPLING OF PCDDs AND PCDFs IN STACK GAS

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ABSTRACT

In an inter-laboratory study the following sampling techniques were compared:

Modified LAGA: Air cooled glass probe, filter, condensate flask and ice cooled impingers.

Cooled probe: Water cooled probe, condensate flask and ice cooled impingers.

Cooled probe-PUFP: Water cooled probe, impinger and polyurethane foam plug.

Dilution method: Stack gases diluted with clean air to below 50°C, filter and florisil adsorbent.

SNV method: Heated glass line and filter, condenser and XAD-2 adsorbent.

Parallel sampling was carried out on two consecutive days at the municipal waste incinerator at Stapelfeld, FRG, and also at the incinerator at Uppsala, Sweden. Intensive studies were undertaken to obtain information about the distribution of PCDDs/PCDFs in all sampling compartments of the techniques used. ^{13}C-labeled PCDDs/PCDFs were introduced into the sampling trains before sampling at various points.

Each sample extract was divided into two equal parts and analyzed at the University of Tübingen and at the University of Umea.

The study shows that all sampling techniques give comparable results if carried out properly. However, the techniques differ considerably in the distribution of PCDDs/PCDFs in the various sampling compartments.

Results obtained in the two laboratories on split sample extracts are in very good agreement.

This study shows that sampling of PCDDs/PCDFs in stack gas can be carried out reliably on a validated basis.

INTRODUCTION

From studies on the comparison of various sampling methods, carried out in 1985 and 1986 by Hagenmaier et al. (1,2,3) the following conclusion was drawn: comparable results are obtained with all sampling procedures (deviations by a factor of 2 were not considered significant). This conclusion was derived, however, only under the condition that losses of ^{13}C-labeled PCDD congeners, added to the sampling trains before sampling was started (sampling spikes), were disregarded. The recoveries for the sampling spikes in these studies were highly erratic.

The Nordic Council of Ministries has recommended that in all Nordic countries a standardized procedure for the measurement of PCDDs/PCDFs in flue gas emissions is used. The important requirements of the recommended protocol are 1) a minimum recovery of 25% of the sampling spikes and 2) the results should be compensated for uncomplete recoveries of the sampling spikes. Investigations by Rappe et al. (4) and Jansson & Bergvall (5) showed that the recoveries for the sampling spikes obtained with the recommended Nordic sampling procedure were below 25% for a high percentage of the samplings (up to 50%).

Low and inconsistent recoveries for sampling spikes on the one hand and comparable results in PCDD/PCDF-concentrations by parallel sampling without compensation for spike recoveries on the other appeared to be an unsatisfactory situation. We therefore started further investigations towards validated sampling of stack gas for PCDDs and PCDFs.

We have previously reported on results obtained in a comparative study conducted at the MSW incinerator at Hamburg-Stapelfeld (6). The results obtained in this study can be summarized as follows. The average recoveries for 8 ^{13}C-labeled PCDDs/PCDFs sampling spikes, added to the sampling train before sampling was started, was on the average (for description of methods see below):

	April 8,1987	April 9,1987
method A	76%	74%
method B	84%	80%
method D	70%	74%
method E	74%	80%

Minimum spike recovery was 47%.

In figure 1 total PCDD/PCDF homologue concentrations for the four sampling procedures are shown for the two sampling periods at Hamburg-Stapelfeld.. The values are corrected for clean-up spikes only. It is obvious that comparable results were obtained for all four sampling procedures. The sample extracts were analyzed both at the University of Tübingen and at the University of Umea. The results obtained in the two laboratories were in very good agreement.

In fall of 1987 this inter-laboratory study was continued with comparative samplings at the MSW incinerator at Uppsala. The results obtained are reported in detail.

EXPERIMENTAL

Sampling procedures compared in this study

At the MSW incinerator at Uppsala, Sweden, five flue gas samples were taken in parallel with five different sampling techniques in two sampling periods. The following sampling procedures were compared:

Water-cooled probe with impingers (7); **method A**
LAGA method 87 (8); **method B**
Water-cooled probe with polyurethane adsorbent (9); **method C**
Dilution method (10); **method D**
SNV-train (11); **method E**

The sampling was carried out by TÜV-Essen (method D), IGU- Hamburg (method A and B) and EMK AB-Malmö (method E) and Institute of Environmental Chemistry, University of Umea (method C).
The MSW incinerator at Uppsala is a mass burning moving grate incinerator with a capacity of 250 000 t/year and an energy production of 90 MW. The incinerator is equipped with an electrostatic precipitator (ESP) followed by a flue gas condensation system. The samples were taken behind the condensation system which was not in operation during the sampling period.

The schematics of the five sampling trains are shown in figures 2-4.

The sampling procedures are characterized as follows:

Method A
A water cooled glass probe ensures a cooling of the sample stream down to below 20°C within seconds. In addition to the cooled probe an ice cooled condensate collector followed by a train of impingers, filled with ethoxy ethanol, are used. The cooled probe is actually a variation of the LAGA 87 procedure, and was developed for PCDD/PCDF-sampling at gas temperatures up to 900°C.
The following sub-samples were collected:

- (a) glass probe
- (b) condensate
- (c) ethoxy ethanol
- (d) washing solvent.

Method B

In contrast to the LAGA 84 train (12) the new German standard procedure is no longer a detailed procedure but rather a recommended method that contains the minimum of demands that the measurement must fulfill. The three major demands are that the sampling probe must be made of glass, that the filter temperature must be below 120°C, and that an absorption train must be used. We complied with these recommendations by using an air cooled glass probe, a filter plug of glass-fiber which showed a temperature of about 60°C, a condensation flask and a train of impingers filled with ethoxy ethanol, both ice cooled.

The following sub-samples were collected:

(a) glass probe
(b) filter
(c) condensate
(d) ethoxy ethanol
(e) washing solvent.

Method C

A water cooled glass probe, identical to the probe used in method A, was used. An ice cooled condensate collecter flask followed by a polyurethane foam plug was connected with the glass probe.

The following sub-samples were collected:

(a) glass probe
(b) condensate
(c) polyurethane foam plug
(d) washing solvent.

Method D

The sample stream is diluted with dry, filtered air in a ratio (e.g. 1:10) that will ensure a temperature at the filter (glass- fiber, 1 m^2, impregnated with paraffin) of <50°C. A back-up of Florisil is used.

The following sub-samples were collected:

(a) filter
(b) cartridge of Florisil adsorbent
(c) washing solvent of probe.

Method E

The Swedish Environmental Protection Board has recommended a sampling technique composed of a glass-fiber filter held at 160°C, followed by a condensor, which takes the temeperature of the flue gas down to at least 20 $^{\circ}$C. The condensate is collected in a flask and the gas passes an adsorbent cartridge containing at least 5 g of XAD-2/m^3 of flue gas sampled.

The following sub-samples were collected:

(a) filter
(b) condensate
(c) XAD-2 adsorbent
(d) washing solvent.

Spiking protocol

Two different spiking protocols were used . On Nov. 4th, one labeled isomer for each degree of chlorination except OCDF was added to each of the sampling trains before sampling was started. On Nov. 5th sampling and clean-up spike were exchanged. In detail the spiking protocol was as follows:

November 4th,1987

Probe spike

25 ng	$^{13}C_{12}$-2,3,4,7,8-Cl_5DF

Sampling spike

20 ng	$^{13}C_{12}$-2,3,7,8-Cl_4DD
20 ng	$^{13}C_{12}$-1,2,3,7,8-Cl_5DD
20 ng	$^{13}C_{12}$-1,2,3,6,7,8-Cl_6DD
20 ng	$^{13}C_{12}$-1,2,3,4,6,7,8-Cl_7DD
20 ng	$^{13}C_{12}$- 2,3,7,8-Cl_4DF
20 ng	$^{13}C_{12}$-1,2,3,7,8-Cl_5DF
20 ng	$^{13}C_{12}$-1,2,3,4,7,8-Cl_6DF
20 ng	$^{13}C_{12}$- 1,2,3,4,6,7,8-Cl_7DF
40 ng	$^{13}C_{12}$-OCDD

Clean-up spike

20 ng	$^{37}Cl_4$-2,3,7,8-Cl_4DD
20 ng	$^{13}C_{12}$-1,2,3,7,8,9-Cl_6DF

Syringe spike

10 ng	$^{13}C_{12}$-1,2,3,4-Cl_4DD
10 ng	^{13}C12-1,2,3,7,8,9-Cl_6DD

November 5th, 1987

Probe spike

25 ng	$^{13}C_{12}$-2,3,4,7,8-Cl_5DF

Sampling spike

20 ng	$^{37}Cl_4$-2,3,7,8-Cl_4DD
20 ng	$^{13}C_{12}$-1,2,3,7,8,9-Cl_6DF

Clean-up spike

20 ng	$^{13}C_{12}$-2,3,7,8-Cl_4DD
20 ng	$^{13}C_{12}$-1,2,3,7,8-Cl_5DD
20 ng	$^{13}C_{12}$-1,2,3,6,7,8-Cl_6DD
20 ng	$^{13}C_{12}$-1,2,3,4,6,7,8-Cl_7DD
20 ng	$^{13}C_{12}$- 2,3,7,8-Cl_4DF
20 ng	$^{13}C_{12}$-1,2,3,7,8-Cl_5DF
20 ng	$^{13}C_{12}$-1,2,3,4,7,8-Cl_6DF
20 ng	$^{13}C_{12}$- 1,2,3,4,6,7,8-Cl_7DF
40 ng	$^{13}C_{12}$-OCDD

Syringe spike

10 ng	$^{13}C_{12}$-1,2,3,4-Cl_4DD
10 ng	^{13}C12-1,2,3,7,8,9-Cl_6DD

Analysis

All sub-samples were analyzed individually. In all cases were a glass probe is given as a sub-sample in the method description the glass probe was cut in pieces (about 10 cm long) after each sampling period. After addition of the clean-up spikes all sub-samples were extracted at the University of Umea and divided into two equal parts. One half of the extract of each sub-sample was sent to the University of Tübingen for clean-up and analysis, the other half was analyzed at the University of Umea. Each laboratory used its own clean-up procedures, described previously

(13,3). At Tübingen a single clean-up step on Alumina B Super I (ICN Biomedicals) was used. Analyses were carried out on a HP5890/HP5970 GC/MSD using a SIL-88 (CHROMPACK) fused silica capillary column for isomer specific analysis of the tetra- to hexaCDD/CDF and a DB-5 (J&W) fused silica capillary column for the analysis of the hepta- and octaCDD/CDF-congeners. At Umea for sample clean-up first a column with Alumina B Super I (ICN Biomedicals) and secondly a column with silica gel treated with H_2SO_4 was used. Analyses were carried out on a SP 2330 (Supelco) fused silica capillary column connected to a high resolution mass spectrometer (VG 70E) running at 5000 dalton in resolution.

For the samples collected on Nov. 4th quantitation was based on the two clean-up spikes and an external determination of response factors using a quantification mixture containing all 2,3,7,8-substituted PCDD/PCDF congeners in a known concentration. For the samples collected on Nov. 5th the quantitation is based on the clean-up spikes added for each degree of chlorination with the exception of OCDF. OCDF was quantified by external standardisation.

On the average the recoveries of the clean-up spikes were with 73.5% identical at both laboratories (Tübingen min. 60.4%, max. 95.6%; Umea min. 42.3%, max. 120.9%). The recoveries reported for the sampling spikes are calculated just for the sampling procedure and are corrected for losses during clean-up.

RESULTS

Comparison of PCDD/PCDF concentrations

In table 1 PCDD/PCDF homologue concentrations and concentrations of 2,3,7,8-substituted congeners obtained with the five sampling procedures and in the two laboratories are shown for the sampling period of Nov. 5th. It is obvious from this table that comparable results are obtained for all five sampling procedures and in the two laboratories.

Distribution of PCDDs/PCDFs in the sampling trains

It is interesting to compare the distribution of PCDDs/PCDFs between the sub-samples for the various sampling methods. This distribution is shown in figures 5 to 9. Especially with the dilution method, a high percentage of the PCDDs/PCDFs is collected in one part of the sampling train, the filter. In the sampling methods which use a water cooled probe the highest percentage of PCDDs/PCDFs is found at least in the condensate, while in the two other procedures the PCDDs and PCDFs are distributed more or less evenly over the whole train and are found to a very substantial amount in the solvent used for rinsing the sampling train. Despite the fact that comparable results were obtained with all five sampling trains destinct differences exist in the efficiency of various parts of the trains to retain the PCDDs/PCDFs. Those methods which yield a high percentage of collected PCDDs and PCDFs in one sampling compartment are certainly better suited for routine measurements than those were the major part is found in the rinsing solvents.

Recovery of labelled sampling spikes

In table 2 and 3 the recoveries of the ^{13}C- and ^{37}Cl-labeled sampling spikes reported from both laboratories are given for the two sampling periods. The recoveries can be considered for all sampling procedures as satisfactory to very good. This shows that proper sampling and sample handling results in good recovery of the sampling spikes. More problematic is the recovery of spikes introduced into the sampling train at some point of the probe (probe spike). As has been observed already in the previous study, these spike recoveries vary to a great extent and are not related to the recovery of the standards placed at other points of the sampling train. The very low recovery of the spikes from the cooled probe (method A) is obviously due to an improper introduction and placement of the spikes in the sampling train. The spikes were introduced into the non-cooled front part of the probe in a toluene solution. The solvent had to be evaporated by heating the glass probe with a heat gun. Spike losses can have occurred therefore either during solvent evaporation or during installation of the probe in the hot stack. This makes evident the importance of a proper introduction of sampling spikes into the sampling train for good spike recoveries. Since this appears difficult to achieve for the introduction of probe spikes, it is not recommended to use these as an indicator for proper sampling and sample handling.

Comparison of toxicity equivalents
In tables 4 and 5 the TCDD-toxicity equivalents are calculated on the basis of the results from the two laboratories for the two sampling periods according to four different methods BGA (14), EPA (15), Eadon (16) and Nordic (17).In table 4 the results are corrected for the recoveries of the clean-up spikes only, in table 5 the results are corrected also for the average recovery of the sampling spikes (without probe spikes). Agreement between the results obtained for the various sampling methods is good with the exception of method E on Nov. 4th, probably due to some technical problems during sampling. Agreement between the results obtained at the laboratories can be considered as good, considering, that for each result a number of sub-samples had to be analysed. The differences in toxicity-equivalents calculated by different methods are greater than the differences between sampling methods and/or laboratories.

CONCLUSION

In summary we can state that it is possible to obtain average spike recoveries consistently better than 50%. Considering however the distribution of the PCDDs/PCDFs and of the sampling spikes between various compartments of the sampling train it becomes obvious that losses are more probable in those trains were the PCDDs/PCDFs and the spikes are more evenly distributed over the whole sampling train than in those in which we find a very high percentage in only one compartment and especially not in the rinsing solution.
It is recommended to use at least two sampling spikes which should be recovered on the average to more than 50%, corrected for clean-up recovery. Good recovery of sampling spikes assures proper sampling and sample handling. In contrast to the Nordic countries in Germany it is not proposed to correct the analytical data for the sampling spike recoveries.

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Figure 1: PCDD/PCDF homologue concentrations for four sampling methods compared at the MSW incinerator at Hamburg-Stapelfeld. Values are corrected for clean-up spike recoveries, but not corrected for sampling spike recoveries.

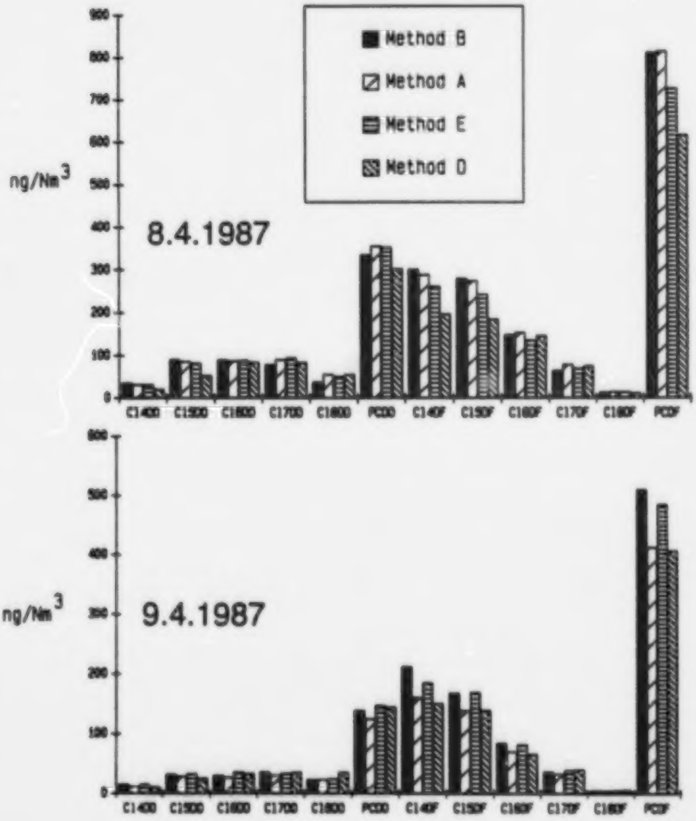


Figure 2: Schematic drawing of the sampling trains of method A and method B (see text). The positions were labeled PCDD/PCDF were introduced before sampling was started are marked with asterisks.

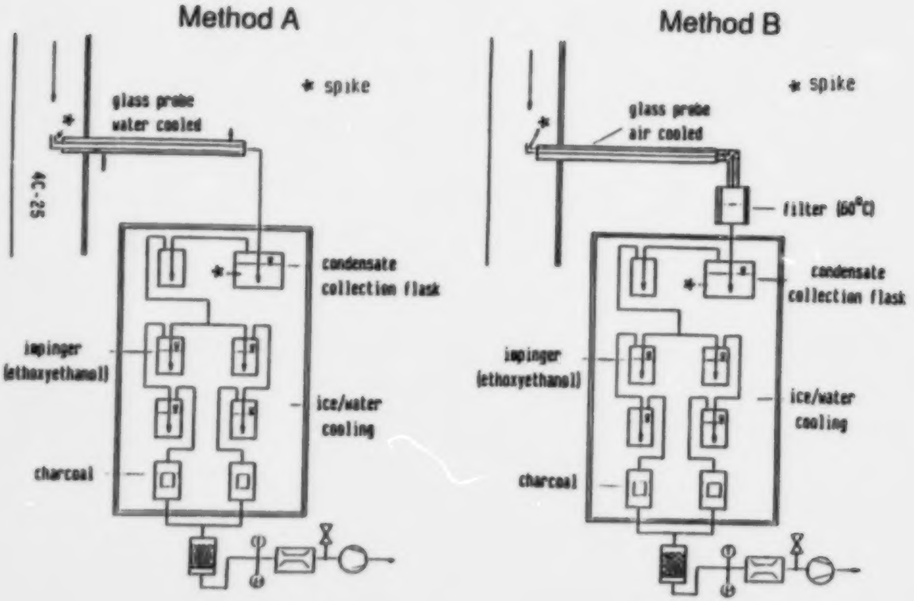

Method A
* spike
glass probe
water cooled
condensate
collection flask
impinger
(ethoxyethanol)
ice/water
cooling
charcoal
Method B
* spike
glass probe
air cooled
filter (60°C)
condensate
collection flask
impinger
(ethoxyethanol)
ice/water
cooling
charcoal

Figure 3: Schematic drawing of the sampling trains of method C and method D (see text). The positions were labeled PCDD/PCDF were introduced before sampling was started are marked with asterisks.

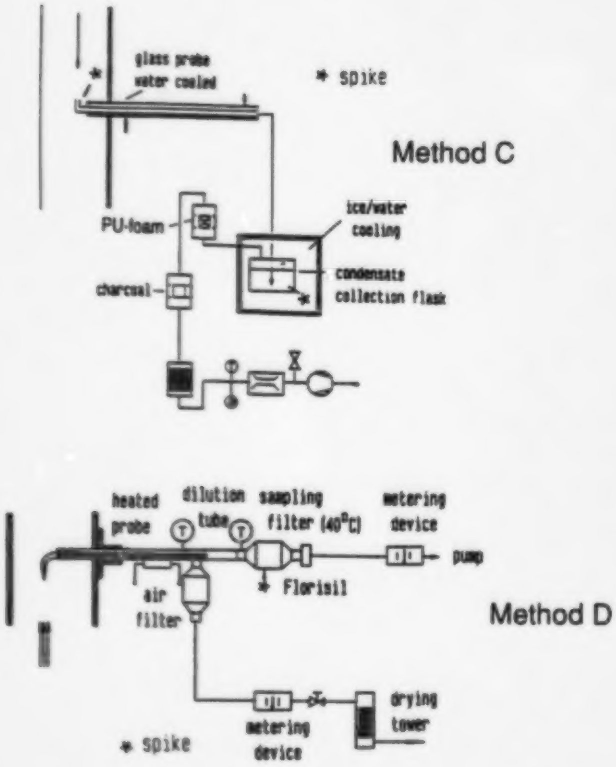
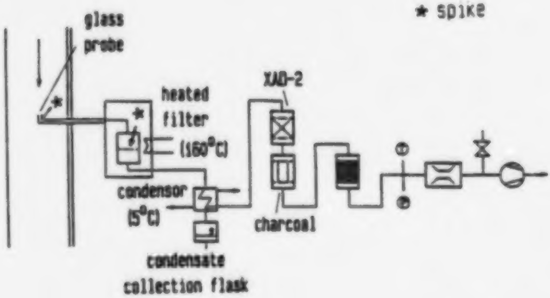

glass probe
water cooled
* spike
Method C
ice/water
cooling
PU-foam
condensate
collection flask
charcoal
*
heated
probe
dilution
tube
sampling
filter (40°C)
metering
device
T
T
pump
air
filter
* Florisil
Method D
drying
tower
metering
device
* spike

Figure 4: Schematic drawing of the sampling train of method E (see text). The positions were labeled PCDD/PCDF were introduced before sampling was started are marked with asterisks.

Method E



★ spike
glass
probe
heated
filter
(160°C)
XAD-2
condensor
(5°C)
charcoal
condensate
collection flask

Figure 5: Distribution of total PCDDs and PCDFs between the various sampling compartments of method A (see text). Comparison of data obtained at the two analytical laboratories in Tübingen and Umea.

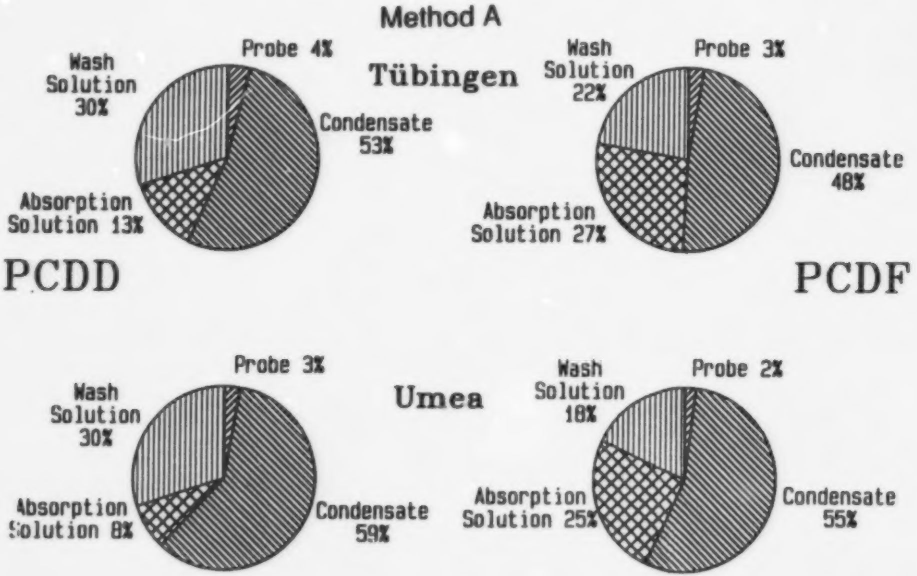


Figure 6: Distribution of total PCDDs and PCDFs between the various sampling compartments of method B (see text). Comparison of data obtained at the two analytical laboratories in Tübingen and Umea.

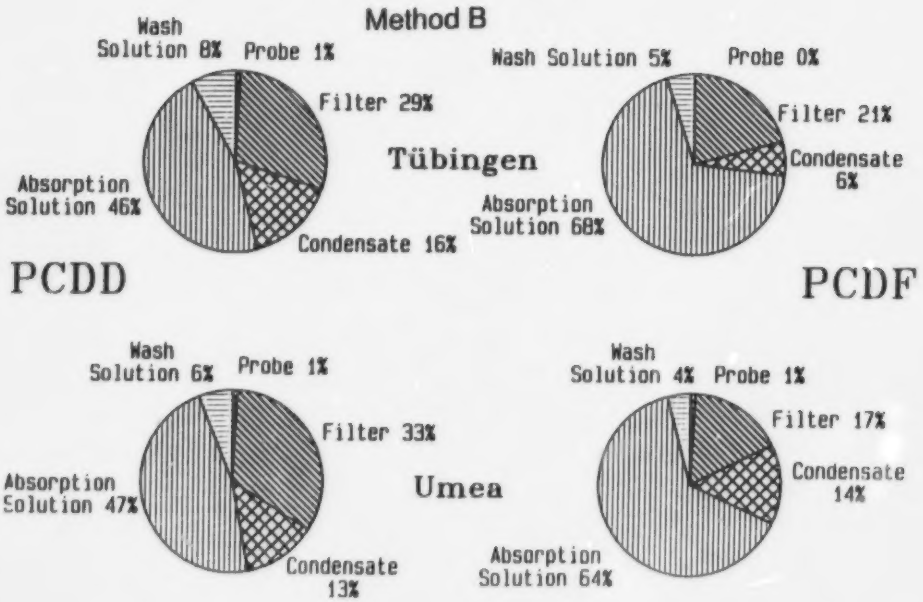


Figure 7: Distribution of total PCDDs and PCDFs between the various sampling compartments of method C (see text). Comparison of data obtained at the two analytical laboratories in Tübingen and Umea.

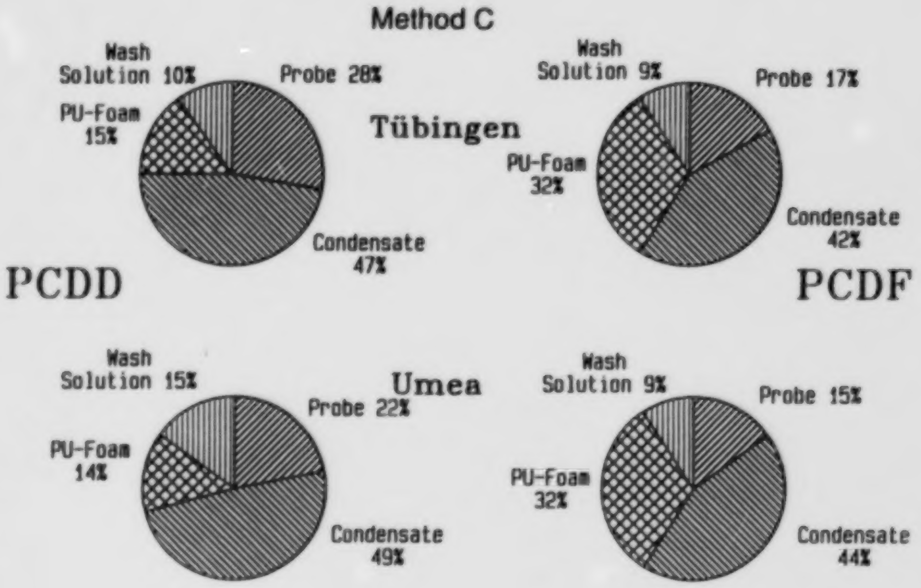


Figure 8: Distribution of total PCDDs and PCDFs between the various sampling compartments of method D (see text). Comparison of data obtained at the two analytical laboratories in Tübingen and Umea.

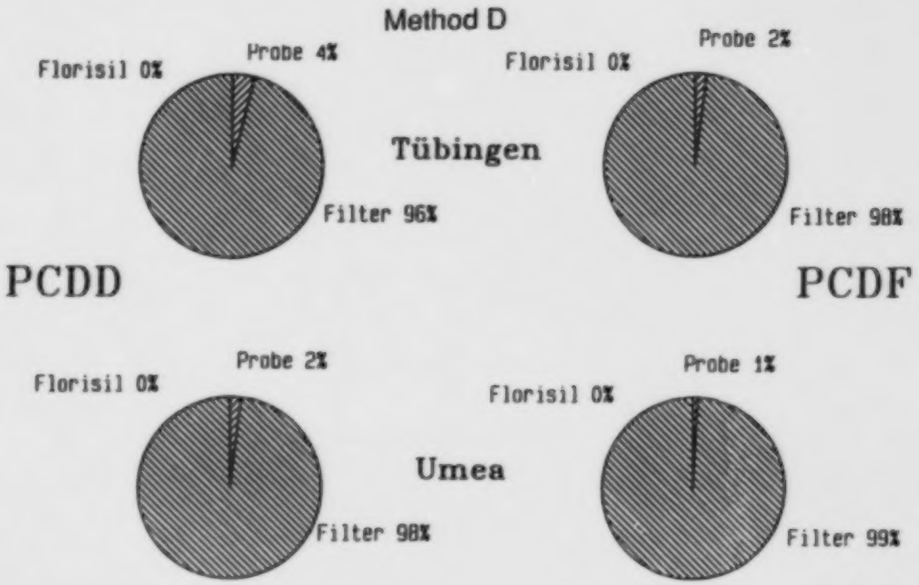


Figure 9: Distribution of total PCDDs and PCDFs between the various sampling compartments of method E (see text). Comparison of data obtained at the two analytical laboratories in Tübingen and Umea.

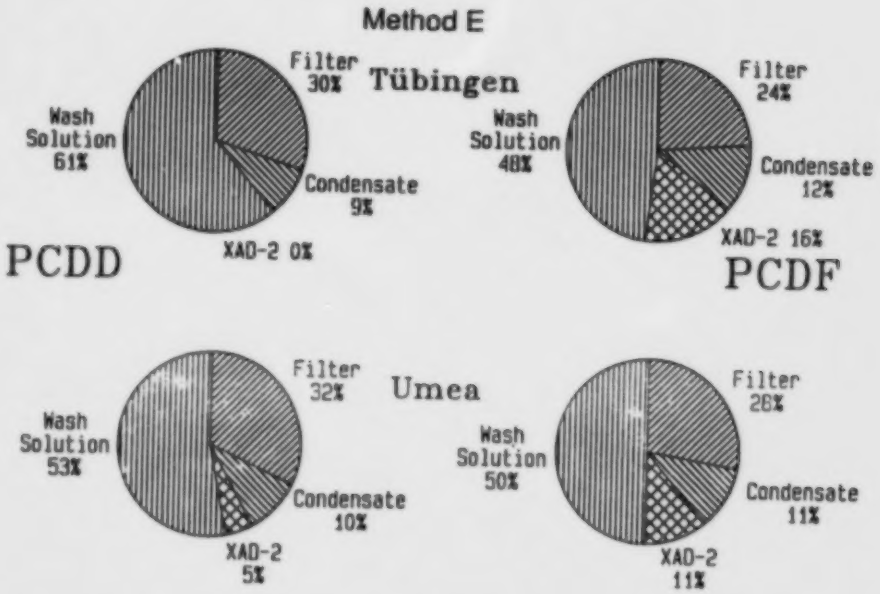


Table 1

Comparison of the analytical results from the University of Tübingen and the University of Umea, for sampling methods A - E. Sampling at the Uppsala municipal waste incinerator (05. 11. 87). Values are corrected for clean-up spike recoveries only. Concentrations in ng/Ncbm. Detection limit for single isomers about 0.01 ng/Ncbm. n.d. = not detected.

	Method A		Method B		Method C		Method D		Method E	
	Tübingen	Umea	Tübingen	Umea	Tübingen	Umea	Tübingen	Umea	Tübingen	Umea
Tetrachlorodibenzodioxins	2.2	2.4	1.9	2.5	2.5	2.6	2.2	2.7	1.1	1.8
Pentachlorodibenzodioxins	3.5	2.9	2.2	2.7	4.0	2.9	4.6	4.3	2.6	2.7
Hexachlorodibenzodioxins	4.8	3.9	4.7	3.8	5.6	4.8	7.6	5.0	4.7	3.1
Heptachlorodibenzodioxins	5.6	6.0	5.4	6.4	6.8	5.8	7.3	11.4	5.8	6.3
Octachlorodibenzodioxin	7.3	4.0	7.7	4.8	9.7	4.9	11.0	6.6	9.1	4.9
Total PCDDs	**23.4**	**19.2**	**21.9**	**20.2**	**28.6**	**21.0**	**32.7**	**30.0**	**23.3**	**18.8**
Tetrachlorodibenzofurans	21.4	24.3	21.4	24.8	21.6	26.7	22.6	29.9	18.0	15.0
Pentachlorodibenzofurans	12.5	13.4	10.8	21.9	11.6	13.5	13.5	17.7	10.5	12.3
Hexachlorodibenzofurans	6.8	13.0	6.2	16.5	6.1	10.9	7.9	15.4	5.6	5.3
Heptachlordibenzofurans	3.9	6.3	4.2	6.2	4.9	5.7	5.4	10.7	1.0	6.1
Octachlorodibenzofuran	1.5	0.4	1.5	0.4	1.5	0.3	3.3	0.6	2.3	0.6
Total PCDFs	**44.1**	**57.4**	**44.1**	**69.8**	**45.7**	**57.1**	**52.7**	**74.3**	**40.4**	**39.3**
2,3,7,8-Tetrachlorodibenzodioxin	0.19	0.16	0.19	0.29	0.15	0.16	0.12	0.20	0.08	0.18
1,2,3,7,8-Pentachlorodibenzodioxin	0.33	0.27	0.29	0.32	0.38	0.24	0.38	0.38	0.30	0.26
1,2,3,4,7,8-Hexachlorodibenzodioxin	0.35	0.24	0.59	0.08	0.61	0.13	0.49	0.25	0.57	0.26
1,2,3,6,7,8-Hexachlorodibenzodioxin	0.49	0.36	0.61	0.17	0.58	0.12	0.66	0.46	0.41	0.32
1,2,3,7,8,9-Hexachlorodibenzodioxin	0.26	0.25	0.20	0.09	0.24	0.13	0.40	0.44	0.18	0.33
1,2,3,4,6,7,8-Heptachlorodibenzodioxin	2.64	2.80	2.72	3.33	3.12	2.12	3.58	5.15	2.90	3.21
2,3,7,8-Tetrachlorodibenzofuran	0.36	0.70	0.42	0.75	0.37	0.71	0.48	0.82	0.35	0.59
1,2,3,7,8-Pentachlorodibenzofuran	0.79	0.95	0.54	1.06	0.80	1.00	0.46	1.44	0.72	0.98
2,3,4,7,8-Pentachlorodibenzofuran	0.51	0.60	0.58	0.66	0.48	0.77	0.68	0.91	0.41	0.72
1,2,3,4,7,8-Hexachlorodibenzofuran	0.70	0.92	0.67	1.27	0.62	1.10	0.79	1.17	0.59	0.91
1,2,3,6,7,8-Hexachlorodibenzofuran	0.71	0.88	0.81	1.13	0.67	0.92	0.82	1.20	0.61	0.79
1,2,3,7,8,9-Hexachlorodibenzofuran	0.03	0.04	n.d.	0.05	n.d.	0.04	0.04	0.18	0.02	0.17
2,3,4,6,7,8-Hexachlorodibenzofuran	0.61	0.47	0.50	1.00	0.57	0.86	0.90	1.54	0.55	0.71
1,2,3,4,6,7,8-Heptachlorodibenzofuran	3.08	3.01	3.49	3.79	3.85	3.64	4.33	5.66	3.15	3.17
1,2,3,4,7,8,9-Heptachlorodibenzofuran	0.14	0.39	n.d.	0.33	0.13	0.19	0.15	0.37	0.07	0.39

Table 2

Total recovery of ^{13}C-labeled PCDD/PCDF spikes from the sampling of Nov. 4th 1987.

Recovery in %

* Probe spike

n.d. = not detectable

	Method A		Method B		Method C		Method D		Method E	
	Tübingen	Umea	Tübingen	Umea	Tübingen	Umea	Tübingen	Umea	Tübingen	Umea
2,3,7,8-Tetrachlorodibenzodioxin	54	38	74	68	74	72	70	73	58	66
1,2,3,7,8-Pentachlorodibenzodioxin	58	32	70	106	71	75	90	66	44	34
1,2,3,6,7,8-Hexachlorodibenzodioxin	52	46	55	73	64	109	85	65	39	65
1,2,3,4,6,7,8-Heptachlorodibenzodioxin	51	50	54	46	74	98	112	50	35	40
Octachlorodibenzodioxin	54	36	54	26	77	99	87	52	39	28
2,3,7,8-Tetrachlorodibenzofuran	62	61	83	67	79	79	85	63	53	48
1,2,3,7,8-Pentachlorodibenzofuran	69	45	88	62	84	79	100	63	57	42
1,2,3,4,7,8-Hexachlorodibenzofuran	52	39	66	89	70	96	98	61	46	73
1,2,3,4,6,7,8-Heptachlorodibenzofuran	59	53	65	56	87	100	95	58	50	61
2,3,4,7,8-Pentachlorodibenzofuran *	n.d.	01	16	19	46	67	----	----	29	25

Table 3

Total recovery of labeled PCDD/PCDF spikes from the sampling of Nov. 5th 1987.

Recovery in %.

* Probe spike

	Method A		Method B		Method C		Method D		Method E	
	Tübingen	Umea	Tübingen	Umea	Tübingen	Umea	Tübingen	Umea	Tübingen	Umea
$^{37}Cl_4$-2,3,7,8-Tetrachlorodibenzodioxin	77	48	113	96	97	49	128	82	92	68
1,2,3,7,8,9-Hexachlorodibenzofuran	44	49	108	89	73	59	116	108	80	101
2,3,4,7,8-Pentachlorodibenzofuran *	52	44	58	80	35	32	----	----	26	27

Table 4
Toxicity equivalents of the comparative study for validated sampling of PCDDs and PCDFs in stack gas. Concentrations in ng/Ncbm. Values are calculated on the basis of data corrected for clean-up spike recoveries.

	Tübingen					4.11.87 Umea				
Toxicity equivalents	Method A	Method B	Method C	Method D	Method E	Method A	Method B	Method C	Method D	Method E
BGA	0.872	0.793	1.295	1.219	0.843	1.314	1.330	1.662	1.185	0.568
EPA	0.449	0.437	0.704	0.617	0.570	0.663	0.756	0.881	0.656	0.289
EADON	0.835	0.836	1.246	1.136	0.928	1.247	1.395	1.674	1.240	0.529
NORDIC	0.699	0.654	1.124	1.073	0.844	1.290	1.211	1.542	1.073	0.499

	Tübingen					5.11.87 Umea				
Toxicity equivalents	Method A	Method B	Method C	Method D	Method E	Method A	Method B	Method C	Method D	Method E
BGA	1.234	1.202	1.228	1.350	1.003	1.318	1.683	1.381	1.869	1.201
EPA	0.662	0.626	0.663	0.646	0.498	0.639	0.832	0.654	0.897	0.661
EADON	1.121	1.031	1.136	1.107	0.921	1.194	1.466	1.252	1.702	1.245
NORDIC	1.036	1.082	1.037	1.208	0.842	1.030	1.321	1.132	1.580	1.149

Table 5

Toxicity equivalents of the comparative study for validated sampling of PCDDs and PCDFs in stack gas.

Values are corrected for the average recovery of sampling spikes.

Concentrations in ng/Ncbm.

	Tübingen				4.11.87	Umea				
Toxicity equivalents	Method A	Method B	Method C	Method D	Method E	Method A	Method B	Method C	Method D	Method E
BGA	1.527	1.144	1.718	1.326	1.767	2.894	1.884	1.878	1.905	1.060
EPA	0.786	0.631	0.934	0.671	1.195	1.460	1.071	0.995	1.055	0.539
EADON	1.462	1.206	1.653	1.236	1.945	2.747	1.973	1.892	1.994	0.987
NORDIC	1.224	0.944	1.491	1.168	1.769	2.841	1.715	1.742	1.725	0.931

	Tübingen				5.11.87	Umea				
Toxicity equivalents	Method A	Method B	Method C	Method D	Method E	Method A	Method B	Method C	Method D	Method E
BGA	2.033	1.086	1.441	1.106	1.164	2.712	1.819	2.539	1.961	1.423
EPA	1.091	0.565	0.778	0.529	0.567	1.315	0.899	1.202	0.941	0.783
EADON	1.847	0.931	1.333	0.907	1.068	2.457	1.585	2.301	1.786	1.475
NORDIC	1.707	0.977	1.217	0.989	0.977	2.119	1.428	2.081	1.658	1.361

HOW SHOULD INCINERATOR ASH BE SAMPLED?

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ABSTRACT

As research into potential options for incinerator ash utilization progresses beyond bench-scale work to field demonstrations, greater attention will need to be given to statistical analysis of the results. In order to make such analyses, procedures need to be followed that will provide for an adequate number of samples. This paper summarizes the basic concepts involved in sampling incinerator ash and in determining how many samples should be taken. A common method of making this determination is described in the USEPA's Test Methods for Evaluating Solid Waste (EPA SW-846). This procedure is illustrated in this paper using data obtained from projects supported by the New York State Energy Research and Development Authority. These illustrations show that using this method can yield a wide range of answers as to how many samples to take. Although a greater number of required samples may be appropriate than the method would estimate, the results of the illustrations indicate that the method can provide a good approximation of the number of samples needed.

INTRODUCTION

Existing and planned municipal solid waste combustors are projected to be producing over 400 megawatts of electricity in New York State by 1996. The incinerator residue remaining after combustion may amount to 2,500,000 tons per year. The energy benefits of these incinerators will not be realized unless satisfactory management alternatives are found for this incinerator residue. Given the shrinking number of available landfilling sites, interest has developed in using ash materials as a resource rather than filling up landfill capacity with it.

The basic factors for evaluating ash utilization systems include market availability, technical feasibility, acceptability of environmental impacts, acceptability of occupational conditions, institutional feasibility and economic feasibility. Sampling and analyses of ash materials is needed to establish the ability of ash utilization systems to meet specified criteria, such as market specifications and environmental and occupational safety standards. This paper is divided into two sections: a review of the basic elements of sampling and an analyses of questions regarding the number of samples to be taken.

BASIC ELEMENTS OF SAMPLING

In general terms, the goal of sampling is to obtain a sufficient number of samples to estimate with an acceptable level of accuracy and precision whether a given quantity of ash material meets specified criteria. The basic elements of sampling to be described in this section include: defining evaluation criteria, choosing physical sampling procedures and estimating the required number of samples. There are a number of other important elements to be incorporated into a good sampling plan such as quality assurance and safety procedures. Descriptions of these important elements can be found in references such as ASTM's Standard Guide for General Planning of Waste Sampling (D4687-87). (1)

DEFINING THE CRITERIA

The first step in achieving this goal is to identify the objective of the analysis to be performed on the samples. In most cases, this will involve identifying specific regulatory standards or market specifications which must be met in order for the proposed ash utilization method to be implemented. The level of probability or confidence in test results which is required to meet the standards or specifications also needs to be specified. For example, the United States Environmental Protection Agency (USEPA) requires the mean of solid waste test results to be less than the appropriate regulatory threshold by at least a 90 percent confidence interval. (2)

BASIC MATERIAL HANDLING ACTIVITIES

This section focuses on the four basic physical activities typically involved in ash sampling: (1) taking simple, uncomposited samples, (2) combining single samples or increments into composite samples, (3) blending composited or uncomposited samples, and (4) taking laboratory-sized subsamples from larger composited or uncomposited samples.

Taking Simple, Uncomposited Samples or Increments

The task of determining the total lead content of a given incinerator's daily residue of some 100 tons of ash would be simple if the ash was homogeneous. Of course this is not the case. For lead content and other test parameters, ash samples have been found to be variable, as would be expected from the heterogeneous mix of waste materials that are burned.

One approach to determining the properties of a heterogeneous material is to analyze all of it. This should accurately determine the lead content of the ash. But considering that 100 tons of ash is enough material for 91 million one-gram total lead analyses, such an approach is obviously impractical. Between the extremes of one and 91 million samples lies a number that should be adequate to show with a specified level of confidence whether the 100 tons meets a given specification. Later sections of the paper will describe how the appropriate number of samples can be estimated. The point to note here is that multiple samples of ash material are needed in order to make useful conclusions about the results.

Compositing of Increments

For a given number of simple samples, the most useful information could be obtained by keeping them separate and analyzing them individually. Analyzing them separately provides information about their variability and might show something about how the ash varies over time.

However, some analyses can cost over several hundred dollars for each sample tested. To keep costs down, individual increments are often combined into composite samples taken over a range of time or location. The resulting composite sample will contain a mix of the characteristics of each of the different increments. Information is lost about how the test parameter may vary over time or location, but the compositing normally smooths out the variability of the individual increments. Thus, fewer analyses should be able to be used to estimate the mean value of the parameter than if the individual increments were analyzed as separate simple samples.

Blending Composited or Uncomposited Samples

The amount of ash material required for chemical or physical analyses is usually less than the amount of material taken for either a simple sample or a composite. Some physical testing, like particle size analysis, may require several kilograms of material. But laboratory leaching tests typically require 100 grams, and organic and total elemental analyses require on the order of 10 and 1 grams of material respectively.

Since only a small portion of a sample is analyzed, it is important that the selected subsample be as representative of the whole sample as practical. Taking a representative subsample from a bucket of ash can be difficult, as was illustrated by the wide range of leaching test results reported by the New York State Department of Environmental Conservation. (3)

The probability of obtaining a representative subsample will increase as the sample is more thoroughly blended. A number of different methods are used to blend samples, including multiple runs through riffles or splitters, grinding with a mortar and pestle, spinning in a tumbler, rolling in barrels, and hand mixing.

Taking Laboratory Subsamples

The methods that can be used for taking a laboratory-sized subsample range from the simple scoop with a spoon to more complex procedures for splitting or coning and quartering. If a composite or simple sample has been very well mixed, then the method used for taking the subsample is less critical. If a scoop off the top is the same as a scoop off the bottom of the sample container, there is no need for elaborate splitting procedures. However, if tests have not been run on multiple subsamples to show that there are no significant differences between subsamples, then more careful subsampling procedures are appropriate.

The specific procedures used for collecting initial increments, compositing, blending and subsampling and then analyzing the ash material in the laboratory, all constitute the physical elements of a specific test method. References 4 through 8 can be consulted for further information about particular procedures that can be incorporated into a specific test method. This paper focuses on the question of how many samples should be taken for a specified test method.

DETERMINATION OF THE NUMBER OF SAMPLES TO TAKE

Determination of the number of samples required to estimate the mean value of a specified ash property can be made according to procedures described in a number of references. The basic procedures typically assume that the test results have a Normal statistical distribution. Although variations exist in these procedures, the calculations generally rest on the same basic factors:

- The degree of accuracy desired.
- The degree of confidence desired.
- A measure of the precision or degree of variability in the method.

The accuracy of a test method can be defined as the degree of agreement between the true mean value of the property being tested and the average of many observations made according to the same test method. Since the test method includes all procedures from the taking of initial samples through laboratory analysis procedures, changes in any of these procedures can result in changes in the accuracy of the results. Note also that the accuracy for each given test method applies only to the specific property being measured. In estimating the number of samples needed, the degree of accuracy that will be acceptable must be defined.

Determining the number of samples to take also requires the selection of a probability factor or a level of confidence in the result that will be acceptable. One ASTM procedure recommends the use of 3 as a probability factor for general use. (9) For Normal distributions, this factor means that the probability of the calculated mean value exceeding the specified accuracy level is only 3 in 1000 or .003. In other terms, using this probability factor yields a two-tailed confidence interval of 99.7 percent that the estimated mean does not exceed the specified error. Different levels of confidence can be obtained by using different factors. A 90 percent one-tailed confidence level can be achieved by using 1.282 instead of 3. USEPA uses this factor when the number of samples being taken is large, but requires larger factors when the number is smaller than 121. The maximum factor is 3.078 which is required when only two samples are being analyzed. Tables of these factors can be found in several references. (1, 2, 9, 10)

The factor representing the precision of the method is essentially the sum of the inherent variation in the properties of the material being analyzed and the error associated with the procedures included in the test method. The precision factor often used is the standard deviation of the test results, which is assumed to represent a practical estimate of the standard deviation of the entire population of possible ash samples.

Since only a small portion of the total population of ash samples is tested and used to estimate the population standard deviation, a range of estimates is to be expected for different samples. If a set of test results includes an extreme value, the resulting sample standard deviation may greatly overestimate the true population standard deviation and the number of samples required to be taken will also be overestimated. On the other hand, if a set of results are all closely clustered, the population standard deviation and the number of samples required might be greatly underestimated.

ASTM Procedure in Recommended Practice E122-72

Several procedures for calculating the required number of samples are described in the Standard Recommended Practice for Choice of Sample Size to Estimate the Average Quality of a Lot or Process (ASTM E 122-72) (9). The most direct is paraphrased here:

The equation for the size, n_r, of the required sample is as follows:

$$n_r = (P*S/E)^2 \qquad (1)$$

where:

E = the maximum allowable error between the estimate to be made from the sample and the result of measuring all the units in the lot or process.

P = the probability factor corresponding to a very low probability that the difference between the sample estimate and the results of measuring (by the same method) all the units in the lot or process is greater than the specified E.

S = the advance estimate of the standard deviation of the lot or process with respect to the test method.

Notes to this formulation describe the need to consider whether the data are normally distributed, since the suggested probability factors are based on a Normal distribution, and skewed distributions can result in different confidence levels than those given.

An example illustrates how this ASTM method is used. Consider a hypothetical situation in which a stockpile of processed ash can be marketed for a coarse aggregate as long as the percent of material that will pass through a 1/2-inch screen is known. Then appropriate calculations can be made for blending the ash material with other aggregate. Assume that the mixing process is able to meet final mix specifications even if the actual average percent passing is off by 10 percent. In other words, if the mean of the samples were to be 78 percent passing the screen, the mixing process could handle the material satisfactorily as long as its true percentage passing was less than 88 percent and greater than 68 percent. Market specifications are assumed to call for a confidence level corresponding to only a one percent probability that the test result will be off by more than 10 percent. Earlier tests using the same method have shown that the results appear to be normally distributed, and that the standard deviations of the previous tests have been relatively close to an average of 15. Since the distribution appears to be normal, the two-tailed probability factor corresponding to 1 percent will be 2.58. Using these factors in Equation 1 results in a required 15 samples in order to be able to conclude that the actual mean value of will be between 68 and 88 percent with 90 confidence.

USEPA Procedure in SW-846

The procedure described in USEPA'S Test Methods for Evaluating Solid Waste, Physical/Chemical Methods (EPA SW-846) is a little more involved but provides a better illustration of a method for comparing test results with regulatory thresholds or other specifications. It can be paraphrased and summarized as follows:

- Determine if the distribution of test results appears to be normally distributed. A simple screen for this is to see if the variance of the results is less than their mean. If not, then the results should be transformed to a normal distribution.

- Determine if the mean of the results exceeds the maximum specification. If so, a different formula will be needed to calculate the number of additional samples needed to show that the ash meets the specification.

- If the mean is less than the maximum specification, calculate the number of samples required, n_r, according to the following:

$$n_r = (t_{(n\ )})*s/(RT-XBAR))^2 \qquad (2)$$

where:

RT = the regulatory threshold or maximum specification

n_i = the number of samples initially or currently analyzed

XBAR = the mean of the n_i samples analyzed

$t_{(n_i)}$ = the probability factor (from Student's distribution) corresponding to the desired low probability that the difference between the estimate of the mean from the n_i samples and the true population mean will exceed the acceptable error represented by difference between the sample mean and the required threshold (RT-XBAR)

s = the standard deviation of the n_i samples analyzed

- Determine if $n_i < n_r$. If true, no conclusion can be made about whether the specification is being met at the specified confidence level, and additional samples would need to be added to bring the total to n_r.

- If $n_i => n_r$, a confidence interval, CI, is calculated which reflects the values of s and $t_{(n_i)}$.

$$CI = XBAR + t_{(n_i)} * s_{\bar{x}} \qquad (3)$$

where:

CI = the upper and lower values of the interval of the tested property beyond which the probability of finding the true mean of the population is less than the probability associated with the selected $t_{(n_i)}$.

$t_{(n_i)}$ = the probability factor defined above.

$s_{\bar{x}}$ = the standard error of the mean, calculated by dividing s by the square root of n_i.

- If the upper limit of the confidence interval is less than RT, the material is determined to meet regulatory or other specifications.

ANALYSIS OF THE SAMPLE NUMBER QUESTION

FURTHER ANALYSIS OF THE EXAMPLE USED IN EPA SW-846

In EPA SW-846, an example is used to illustrate the calculation of confidence intervals and the number of samples required to show that a solid waste sample meets a regulatory threshold (RT). This data can be further analyzed to illustrate the need to be careful about making conclusions based on small data sets.

The hypothetical example uses a set of barium concentrations for which the regulatory threshold (RT) is 100 parts per million (ppm). The following nine test results are analyzed: 89, 90, 87, 96, 93, 113, 93, 90, and 91 ppm. The mean of 93.56 is less than the RT, but the upper limit of the confidence interval (CI) must be determined to see if it can be concluded that the material will not exceed the RT. The variance of the samples is 60.03, less than the mean. Thus, it is presumed that the distribution is normal and the upper CI limit is calculated using Equation 3 with the probability factor appropriate for a one-tailed 90 percent confidence interval with (9-1) degrees of freedom. The resulting upper limit is 97.16. As determined by Equation 2, the number of samples required to set the 90 percent confidence limit is 2.4, or 3 to the next whole number, which is less than the nine sample results that had been analyzed. It is then concluded that barium is not present at a hazardous level.

Since n_1 was determined to be 2.4, or say 3 tests, a question that might be asked is whether this "non-hazardous" conclusion might have been reached by taking only the three tests indicated instead of the nine.

In order to examine this question more carefully, different subsets of the nine sample results were analyzed to see what confidence interval limits and estimates of required future sample numbers would have been calculated had these different subsets been used. To do this analysis a computer model was developed that calculated the subset means, required number of samples and upper confidence interval limits for all possible subset combinations of the nine samples taken 2, 3, 4, 5, 6, 7 and 8 at a time. The model followed the SW-846 procedure as previously described but used the square root transformation for all subsets in which the variance was greater than 95 percent of the mean. The arcsine transformation, which is suggested in EPA SW-846 for cases in which the variance exceeds the mean, did not appear appropriate for the small number of samples involved. The variances of all transformed subsets was less than their individual means, indicating that the square root transformation successfully transformed the data into a relatively Normal distribution.

As summarized in Table 1, the results show that the estimate of the number of samples required can vary greatly depending on the combination selected, and that these estimates can be quite different from the required sample size of 3 that was calculated based on the full complement of nine samples. The results also indicate that this inadequacy may not be as serious as it first appears.

The combination analyses show that a wide range of required sample size estimates can result. An unlucky sampling team that had only taken the worst three samples of the nine, would have calculated that 1,153 samples would be needed to show that the material was below the RT. Other sets of three samples would indicate that one sample would be adequate. Only 67 percent of the 84 possible sample combinations of the nine samples taken three at a time were found to meet the RT. This contrasts to the 90 percent that might have been expected based on the selected probability factor.

Evaluating the number of samples and percent passing for all the possible combinations of four and five samples showed that the maximum estimated sample number required significantly decreased. This was expected since the maximum variance was reduced due to a larger sample size (4 or 5 versus 3). It was somewhat surprising, however, to find that the percentage of combinations passing decreased from the 67 percent shown for the three sample combinations, to 56 and 44 percent for the four and five sample combinations, respectively.

Even though the analysis of the full complement of nine data points indicates that three samples should be sufficient to set the 90 percent confidence limit below the RT, as illustrated in the graph for the different subsets of these nine, the percentage of combinations passing does not reach 90 percent until 7 samples are analyzed. These are more samples than the estimated three, but seven samples is much less than the 1,153 samples that some unlucky sampling team might have found by taking the worst combination of three samples. Even though initial sampling efforts may find that the number of samples subsequently required to meet a specification is extremely high, the data in Table 1 indicates that it may be possible with only a few more samples to greatly reduce the number required. Calculations of the number of samples required based on a relatively few samples should be confirmed with additional sampling events.

ANALYSIS OF PARTICLE SIZE DATA

As part of the work of ash utilization projects sponsored by the New York State Energy Research and Development Authority, contractors have been asked to devote some of their effort to developing information about how variable ash is. This paper covers an initial analyses of such data.

Description of the Data

A sizeable data set on bottom ash particle size has been gathered by Herb Tucker of Kinetics Technology International from the ash generated by a 36 ton-per-day Clean Air modular unit operating in Cattaraugus County, New York. Part of the initial sampling effort was devoted to developing a method that would indicate how well the sampling team was doing in collecting a sample typical of the ash being generated. One parameter that seemed relatively easy to determine and which might give some measure of ash variability is the particle size distribution.

Random sampling from a grid was the first sampling method tried. Six random samples were collected from a 30-cell grid set over a layer of ash that was spread out in the bottom of a rectangular ash container to a depth of about nine inches. The individual samples were screened in the field into four size fractions using 1-inch, 1/2-half and 1/4-inch screens. After the samples were collected, the remainder of the ash in the container was also screened, so that the true particle size distribution of all the ash in the container could be determined. The weights in the four fractions of ash material are given in Table 2.

The second method consisted of collecting samples with a bucket loader positioned under a conveyor drop approximately every 12 minutes for about three hours. Twelve samples were collected on each of four days. Each sample was separated into three fractions in the field using 1-inch and 1/2-inch screens. The weights of these fractions are provided in Table 3.

One parameter common to the two sampling methods is the percentage of minus 1-inch material that also passed the 1/2-inch screen. This parameter was chosen for calculating the sample means and variances which are shown in Tables 2 and 3. The conveyor drop method showed fairly consistent results in each of the four days with means ranging from 81.1 to 80.2 percent. This showed good agreement with the 81.0 percent found from the earlier screening all 1,556 pounds of ash in the grid container. The average of the six grid method samples of 78.3 underestimates the actual contents of the minus 1/2-inch fraction in the container.

The variance of the conveyor drop samples for each day were less than each day's mean, indicating a more normal distribution than the grid samples which showed the variance exceeding the mean. The grid method was also found to be less desirable in that there was a tendency of different particle sizes to stratify as they fell into the container. This problem was avoided in the conveyor drop method where all materials dropping off the belt were collected.

Another set of particle size data was collected at a large scale, mass-burning unit as part of the Energy Authority's project with the Long Island Regional Planning Board. A hand shovel was used to collect the samples approximately every 20 minutes from the unit's bottom ash conveyor. As in Cattaraugus County, the material was then screened in the field using some coarse screening sizes. The weights of the screened fraction are shown in Table 4.

The percent of minus-one inch material which also passed the 1/2-inch screen is also shown in Table 4. The mean of 76 percent was similar to the 80 to 81 percent found in the daily samples taken at Cattaraugus County. However, the variance of this fraction was found to be appreciably below that found in the Cattaraugus County testing. Further testing would be needed to determine if this resulted from differences in the sampling procedures or in the ash itself.

Calculations of the Number of Samples Required

The data from the different sampling events were also analyzed to see what conclusions might be reached about the number of sample required. For purposes of comparison, the analyses are based on the same confidence level set by USEPA in SW-846, i.e., 90 percent for a one-tailed probability. For purposes of analyzing the particle size distribution a potential maximum market specification of 85 percent is assumed for the percent of minus 1-inch material passing the 1/2-inch screen.

Using the procedure summarized above from EPA SW-846, it was determined that the grid sample and all the conveyor samples could meet the 85 percent specification. In all cases, the upper limit of the 90 percent confidence interval was less than 85 percent, and the number of samples taken was less than the estimated number required.

However, hand screening of the hundreds of pounds of ash involved in the conveyor method required a significant effort. The question naturally arises as to whether good enough results could be obtained without such a labor intensive effort. Analysis of the first day's 12 samples at Cattaraugus County indicated that four samples might be sufficient in subsequent days to meet the 85 percent specification with a 90 percent confidence interval. To examine this estimate, the computer model described in a previous section was used to calculate all the possible four sample subsets of each day's 12 samples. The results are summarized in Table 5. For day four, all 495 possible combinations would have met the 85 percent spec, but only 27 percent and 57 percent would have met it for the second and third days respectively. The number of samples calculated to be needed using all these combinations of four samples ranged from 1 to 75,539 for day two and from less than 1 to 7,233 for day three. Fortunately, there are only a relatively small number of combinations with extremely high values for the required sample numbers. The extremely high numbers result from "unlucky" combinations with a high variance.

As shown in Table 5, the percentage of different combinations of each day's conveyor samples that met the 85 percent specification varied significantly from day to day over the range of sample numbers contained in the subsets. For the Cattaraugus County data, as with the barium data analyzed in an earlier section, the number of samples needed appears to be a little bit larger than equation 2 would indicate. And as also found above, extremely high numbers of required samples may well be reduced to more manageable levels with the addition of a few more samples that reduce the large variance caused by extreme values that are not representative of the ash as a whole, but which can result from an "unlucky" solution of samples.

The analysis of the 12 conveyor samples from the larger, mass burning unit did not show that a large number of samples would be required. This is consistent with the fact that the mean of these 12 samples was further below the 85 percent specification and that their variance was somewhat less than the daily sets of data from Cattaraugus County.

CONCLUSIONS AND RECOMMENDATIONS

Data has been gathered on the particle size distribution of incinerator ash samples taken using different sampling methods. Based on preliminary analyses of one particle size fraction, it appears that collecting material using either of the two conveyor methods may prove more accurate and precise than the grid method.

The results showed that the conveyor sampling methods yielded less variability of results, and mean values that appear to more closely correspond to the actual percentage of minus 1/2-inch material found in the bottom ash. These preliminary conclusions are based on limited sampling, particularly using the grid method. Further sampling would be needed to make a more definitive comparison.

Data from the sampling events was used to calculate the number of samples that would be required for the given sampling and testing methods to meet specified particle size specification with a 90 percent confidence level. The results illustrate that the calculated sample size can vary widely when based on small numbers of initial samples. If the initial samples chosen have a high variance due simply to chance, the number of required samples can be exceedingly high. However, the results do indicate that taking a few more samples might well result in a significant reduction in the number of samples required.

The analyses indicate that initial estimates of the number of samples required should be used with caution, particularly when based on a small initial sample size. Where possible, ash sampling and testing programs should include an allowance for additional testing that may be needed to gain a more reliable understanding of the variance of the sampling and analysis methods being used.

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TABLE 1. RESULTS OF SUBSET ANALYSES
OF NINE DATA POINTS FROM SW-846

Samples Per Subset	Number of Possible Subsets	Percent of Subsets With CI<85%	Maximum Required Number of Samples
2	36	64	6
3	84	67	1153
4	126	56	158
5	126	44	24
6	84	54	10
7	36	100	6
8	9	100	4
9	1	100	3

TABLE 2: SCREENING RESULTS FROM GRID SAMPLES OF CATTARAUGUS COUNTY BOTTOM ASH

Grid Number	Weight in Each Fraction in pounds (wet weight) +1 inch	-1 inch +1/2 inch	-1/2 inch +1/4 inch	-1/4 inch	Percent -1/2 inch of all -1 inch material	
22	2.50	2.88	9.88	22.50	91.83	
9	14.44	10.00	12.88	20.75	77.08	
24	1.75	4.25	9.44	18.31	86.72	
21	6.63	10.63	15.00	24.13	78.64	
2	36.50	4.88	5.25	8.75	74.15	
26	14.00	10.19	6.31	10.06	61.63	
Remainder	270.58	186.93	261.47	555.45	81.38	$\bar{x} = 78.36$[a]
Total	346.40	229.76	320.23	659.95	81.01[b]	$s^2 = 110.67$[c]

[a] average of 6 samples (unweighted)

[b] average of total weights of each fraction

[c] variance of the 6 unweighted samples

TABLE 3. SCREENING RESULTS FOR CATTARAUGUS COUNTY BOTTOM ASH FROM CONVEYOR DROP

DAY 1					DAY 3				
Pounds Wet Weight			Percent		Pounds Wet Weight			Percent	
+1 inch	-1 inch +1/2 inch	-1/2 inch	-1/2 inch of all -1 inch material		+1 inch	-1 inch +1/2 inch	-1/2 inch	-1/2 inch of all -1 inch material	
22.00	23.75	84.25	78.01		5.00	6.50	34.00	83.95	
8.00	5.50	12.75	69.86		7.50	5.50	20.00	78.43	
34.00	17.75	60.25	77.24		7.75	5.00	24.75	83.19	
7.50	6.00	34.00	85.00		5.00	4.00	25.75	86.55	
5.50	2.25	18.50	89.16		12.50	8.00	34.25	81.07	
20.00	13.00	52.00	80.00		6.75	5.25	21.75	80.56	
3.00	5.75	32.00	84.77		21.50	5.75	25.50	81.60	
26.00	15.00	58.00	79.45		6.75	5.50	20.75	79.05	
9.00	6.00	33.00	84.62		7.50	8.25	26.50	76.26	
22.00	16.00	76.00	82.61		11.75	8.75	22.00	71.54	
13.50	6.50	37.00	85.06		3.50	4.00	28.50	87.54	
25.00	14.00	53.00	79.10	$\bar{x}_2$= 81.24[a]	20.25	13.00	54.50	80.74	$\bar{x}_2$= 80.89[a]
195.50	131.50	550.75	80.73[b]	s^2 = 25.68[c]	115.75	79.50	338.25	80.97[b]	s^2 = 19.26[c]
DAY 2					DAY 4				
7.50	6.00	10.50	63.64		5.00	10.25	33.25	76.44	
10.25	10.25	42.75	80.66		19.00	7.75	41.00	84.10	
8.00	6.00	28.00	82.35		8.25	5.00	24.25	82.91	
6.25	5.50	20.50	78.85		19.00	5.00	25.25	83.47	
7.75	3.00	20.00	86.96		10.50	14.75	50.75	77.48	
13.25	5.00	24.00	82.76		14.00	4.75	21.50	81.90	
4.00	3.00	27.50	90.16		18.00	9.00	36.00	80.00	
13.00	6.00	25.00	80.65		5.00	7.75	23.00	74.80	
3.75	4.00	30.50	88.41		7.50	6.00	31.00	83.78	
11.00	6.75	38.00	84.92		16.75	12.00	41.25	77.46	
24.75	13.25	43.50	76.65		9.75	5.00	22.00	81.48	
23.75	13.50	45.50	77.12	$\bar{x}_2$= 81.09[a]	6.75	7.00	24.75	77.96	$\bar{x}_2$= 80.15[a]
133.25	82.25	355.75	81.22[b]	s^2 = 48.61	128.75	94.25	374.00	79.87[b]	s^2 = 10.35

[a] average of 12 samples (unweighted)
[b] average of total weights of each fraction
[c] variance of the 12 unweighted samples

TABLE 4: SCREENING RESULTS FOR MASS BURNING FACILITY BOTTOM ASH SHOVELED FROM CONVEYOR

Pounds Wet Weight			Percent		
+2 inch	-2 inch +1 inch	-1 inch +1/2 inch	-1/2 inch	-1/2 inch of all -1 inch material	
5.69	4.75	7.25	28.75	79.86	
5.37	5.94	7.25	21.50	74.78	
4.75	3.75	7.13	21.12	74.76	
3.44	3.50	9.25	27.50	74.83	
1.75	3.63	9.87	30.50	75.55	
5.00	3.88	9.62	28.25	74.60	
2.00	3.25	7.25	24.75	77.34	
2.50	3.75	9.25	31.75	77.44	
8.25	2.25	6.75	21.25	75.89	
12.75	3.75	7.00	20.75	74.77	
4.31	3.69	7.50	21.50	74.14	
1.50	3.50	6.75	30.00	79.47	$\bar{x} = 76.12^a$
57.31	45.64	94.87	307.62	76.43^b	$s^2 = 3.83^c$

[a] average of 12 samples (unweighted)
[b] average of total weights of each fraction
[c] variance of the 12 unweighted samples

TABLE 5. RESULTS OF SUBSET ANALYSES OF BOTTOM ASH CONVEYOR SAMPLES FROM CATTARAUGUS COUNTY UNIT AND LARGER MASS BURNING UNIT

Samples Per Subset	Number of Possible Subsets	Percent of Subsets With CI <85%				
		Cattaraugus County Unit				Larger Mass Burning Unit
		Day 1	Day 2	Day 3	Day 4	
2	66	18	23	30	36	89
3	220	34	25	50	85	100
4	495	49	27	57	100	100
5	792	58	36	80	100	100
6	924	68	45	88	100	100
7	792	81	54	99	100	100
8	495	94	66	100	100	100
9	220	100	76	100	100	100
10	66	100	91	100	100	100
11	12	100	100	100	100	100
12	1	100	100	100	100	100

FLUID-BED ENERGY RECOVERY FACILITIES: OPERATING PRINCIPLES AND ENVIRONMENTAL PERFORMANCE

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ABSTRACT

With conventional combustion technologies for waste-to-energy, solid waste is burned on the surface of a grate or hearth, or in a shallow suspension, just above the grate surface. With fluid-bed technology, however, solid waste is burned entirely in suspension, with no grate or hearth present at all. It is this suspension that is referred to as the "fluid bed," a highly turbulent mixture of solid waste and super-heated inert "bed material," normally comprised of particles of sand and ash residue.

While fluid-bed technology is well-accepted today for many industrial applications, only recently has the technology received serious consideration in the United States for municipal waste-to-energy applications. In this paper, the features of fluid-bed technology that distinguish it from conventional combustors for waste-to-energy are briefly reviewed. The operational and environmental attributes of fluid-bed technology are then addressed. Finally, stack emissions test data, including dioxin data, are presented for the one refuse-fired fluid-bed facility operating in the United States, and for such facilities operating in Scandinavia. In addition, toxicity test data for ash residue are presented.

FLUIDIZED-BED COMBUSTION TECHNOLOGY

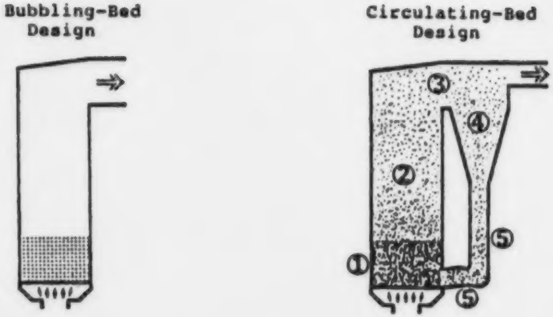
Below, the operating principles of fluidized-bed combustion are described, as compared with solid-waste or refuse-derived fuel (RDF) combustion using conventional grate- or hearth-type incinerators. The claimed attributes of fluidized-bed technology for solid waste combustion are then discussed.

PRINCIPLE OF FLUIDIZED-BED COMBUSTION

With conventional, mass-burn incinerators operating in the United States today, the solid waste is combusted on the surface of a grate or hearth, near the bottom of the furnace. Convective agitation of the waste during combustion is relatively minimal, and is in fact, normally aided by mechanical means. With conventional RDF-fueled incinerators, RDF is "blown in" over the grate, with the RDF being burned in a shallow suspension above the grate surface, and the burn-out being completed on the grate itself.

Fluidized-bed incinerators do not burn the solid waste on a grate or hearth. Rather, the waste is burned in a hot, turbulent suspension of material, entrained in a large upward flow of primary combustion air. This fluid suspension, the "fluidized bed," consists of intensely heated particles of an inert material, intimately mixed with a relatively smaller amount of solid waste being combusted. The inert material is comprised of particles of sand as well as ash from the burned waste. Limestone particles may also be added to serve as a portion of the fluidized bed.

Fluidized bed technology is illustrated in Figure 1 for two types of fluid-bed combustors: "bubbling" fluidized-bed technology (BFB) and "circulating" fluidized-bed technology (CFB). In the bubbling-bed design, a hot, dense fluid mass is maintained near the bottom of the furnace. The effect of the fluidizing of the inert bed particles is similar to the behavior of corn kernels being heated in a hot-air popcorn popper. With the CFB design, there is a similar, dense, lower zone of bed material [see position (1) in Figure 1], but in addition, a less-dense upper zone (2). Primary combustion air is supplied to the bottom of the dense zone of the fluidized bed, and secondary air to the less-dense upper zone. With CFB combustion, a fraction of the bed material then leaves the furnace, entrained in the combustion gas (3), and enters a cyclone separator (4). This cyclone recovers



Bubbling-Bed Design

Circulating-Bed Design

FIGURE 1: FLUID-BED COMBUSTION

the majority of bed material, and the recovered material is then returned continuously (5) to the dense bed, thus the name "circulating fluidized bed."

Examples of operating BFB plants that combust municipal solid waste are the Duluth facility, and also facilities in Japan such as those operating at Koza and Machida. Examples of CFB plants fueled with municipal waste are the Sundsvall Plant in Sweden and the Sande Plant in Norway.

ATTRIBUTES OF FLUIDIZED-BED COMBUSTORS

BFB is a simple, efficient combustor design that is ideally suited for single-fuel combustion (e.g. RDF), in applications requiring an ordinary, relatively-low turn-down ratio (typical BFB turn-down ratio is 1.5:1). To illustrate the concept of turn-down ratio, a high (i.e. favorable) turn-down ratio would be 4:1, which means the boiler can operate at one-quarter of full capacity, if desired, without major loss of boiler efficiency.

CFB is a more advanced technology that provides increased flexibility in the combustion process (e.g. for multiple-fuel combustion). The attributes claimed for both the BFB and CFB technologies are:

- High Combustion Efficiency -- Recalling the "three T's" of efficient combustion (temperature, turbulence, and time), fluidized-bed combustors demonstrate superior combustive turbulence. The fluidized bed provides an intensely-turbulent, intimate mixture of fuel, hot inert bed particles and air. A combustion efficiency exceeding 99% (i.e. less than 1% unburned carbon) is claimed(1) for fluidized-bed combustors. By comparison, conventional solid-waste combustors achieve a combustion efficiency of 97% to 98%.

- More Stable Combustion -- due to a "thermal flywheel effect." A tremendous amount of heat is absorbed and retained by the large mass (i.e. the "inventory") of inert particles making up the fluid bed. The heat-radiating surface area of the inert sand and ash making up the fluid bed is approximately ten times the radiating surface of the furnace itself. The large thermal mass and extreme turbulence greatly reduce the potential for cold and hot spots to occur within the furnace, in turn reducing the potential for stratified pockets of poor combustion to occur.

- Lower Combustion Temperatures -- Efficient combustion is achieved with fluidized-bed technology at lower combustion temperatures (1500-1700°) resulting in reduced formation of oxides of nitrogen (NO_x). NO_x emissions are typically in the range of 150-200 ppmv.(2) This compares with NOx concentrations of 200-350 ppmv typically achieved with conventional combustion. The lower temperatures also reduce the potential for ash slagging in the furnace.

- Desulfurization -- removal in the furnace ("in situ") of some sulfur dioxide (SO_2) from the combustion gas by addition of granular limestone to the bed material. Based on test results for BFB combustion of low sulfur (.3%) coal, SO_2 removal efficiency on the order of 40-50% with in situ scrubbing can be expected.(3) While this removal efficiency range may be sufficient to meet SO_2 emission limits for combustion of some fuels, a flue gas scrubber may be necessary with BFB to guarantee the removal efficiencies normally mandated for solid-waste combustion, i.e. 70-75% SO_2 removal, and 90% acid-gas removal.

The additional advantages claimed for CFB technology are as follows:

- Multi-Fuel Capability -- The CFB boilers are able to burn waste fuels of relatively low fuel quality such as RDF, as well as fuels of higher quality such as tires. This is because the degree of heat extraction from the boiler can be adjusted by varying the density of bed material from the lower to upper furnace zones. For example, in burning a high quality fuel such as tires, bed material would be shifted to the top of the furnace, increasing heat transfer there, and providing the greater degree of heat extraction required overall from the furnace to burn such fuel.

- Improved Combustion Efficiency -- High combustion efficiency is important in minimizing dioxin emissions. The turbulent mixing and thermal-fly wheel effect characteristic of all fluidized bed boilers are further enhanced in the CFB boiler due to the ability described above to optimize the density of the recirculating bed material at any zone in the furnace volume. The resulting vigorous mixing of combustibles with air, the uniform combustion temperature within the entire furnace volume (varies by no more than ± 50°F anywhere within the furnace volume), and the long residence time (5 seconds) of the gases in the furnace make for very high combustion efficiency. The 5-second gas residence time in a CFB furnace is three to four times the gas residence time of a conventional furnace (1-2 seconds).

- Improved Boiler Efficiency -- CFB technology can achieve a boiler efficiency of 80%, or more, due to low excess air requirements (30%) and low unburned carbon (1% or less). This compares with typical boiler efficiencies of 70% for conventional combustors that burn waste that has not been preprocessed, and 75% for conventional RDF combustors.

- Improved Turndown Capacity -- Because of the ability to control the heat-extraction rate, CFB boilers can operate at any load between 25% to 100% capacity, with little change in combustion gas temperature.

- Improved NO_x Suppression -- Besides operating at relatively low combustion temperatures (1500°F to 1700°F), the CFB boiler operates with staged combustion (i.e. the lower and upper combustion zones). The lower zone, therefore, can be operated in a reducing atmosphere, and the upper zone in an oxidizing atmosphere. The lower furnace temperatures and the reducing atmosphere of the first furnace combustion zone suppress NO_x formation.

- Efficient In-Furnace Scrubbing -- With injection of granular limestone along with the waste into the CFB furnace, the intense combustive turbulence, long gas residence time, and high bed-circulation velocity cause intimate mixing of the limestone sorbent particles with the combustion gases.
  Highly-efficient scrubbing of SO_2 and acid gases -- hydrogen chloride (HCl) and hydrogen fluoride (HF), are possible in the furnace as a result. Based on initial test data presented later in this paper, CFB boilers demonstrate the potential to achieve SO_2

and acid-gas scrubbing efficiencies in situ that are as high as those achieved by modern flue-gas scrubbers used with the conventional solid-waste combustion technologies.

U.S. FACILITIES -- OPERATING AND PLANNED

WLSSD FACILITY, DULUTH, MINNESOTA

Only one fluid-bed energy-recovery facility has been built in the United States to-date that uses municipal solid waste as fuel, and also operates 24-hours per day. This facility is operated by the Western Lake Superior Sanitary District (WLSSD), and combusts RDF and de-watered sewage sludge. Plant design information is summarized in Table 1.(4)

The history of this plant is interesting. After going on-line in 1982, the facility operated only several months before being shut down because of glass-slagging problems. Following redesign of the RDF process to remove a higher fraction of glass and metal, the facility began operations again in mid-1985, processing RDF, sewage sludge, and wood chips. The facility has operated very reliably since 1985. On-line availability of the RDF process and fluid-bed facility have been near 95% and 85%, respectively, since June 1986.(4)

ERIE, PENNSYLVANIA AND ROBBINS, ILLINOIS FACILITIES

At least two, large municipal waste-to-energy facilities currently planned in the United States will use CFB combustors. These RDF-fueled plants will be privately developed by the Reading Energy Group of Philadelphia, Pennsylvania. The first plant is to be constructed in Erie, Pennsylvania. Construction permits were issued for this facility in 1987, and construction will begin after resolution of a pending permit appeal. The second CFB facility is planned for Robbins, Illinois. Permit applications are under agency review at present, and construction is expected to commence in 1989.

Both the Erie and Robbins facilities will use Gotaverken CFB boilers, designed in Sweden. The same Gotaverken CFB boilers are used at the Sundsvall resource-recovery plant in

TABLE 1: DESCRIPTION OF WLSSD OPERATING BUBBLING-BED FACILITY

Facility Name/ Location	WLSSD Facility, Duluth, MN
Public Opertor	Western Lake Superior Sanitary District (WLSSD)
Fuels	RDF (4" max.), de-watered sludge
Supplemental Fuels	Fuel oil, wood chips
Combustor Type	Bubbling-Bed (Copeland)
Combustor Capacity	Two units @ 120 TPD RDF, plus 345 TPD sludge each[a]
Bed Operating Temperature	1475°F
Boiler Type	Waste-Heat Boiler (Deltak)
Steam Production	49,000 lb/hr, each of two boilers
Energy Recovery	Steam, used by the WLSSD Wastewater Treatment Plant
Air Pollution Controls	Cyclone, venturi scrubber, tray scrubber, demister
Status	Operating

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[a] Only one combustion unit operates at a time at the WLSSD facility.

Sweden and at the Sande Plant in Norway, which are fueled with RDF, wood chips, and other biomass fuels, and have demonstrated successful commercial operation for four years. For example, Gotaverken indicates that the Sundsvall Plant CFB boilers have achieved an on-line availability of 86% since commercial start-up in October 1984.

Design information for the planned Erie and Robbins facilities is summarized in Table 2. Note from Table 2, that at both the Erie and Robbins CFB facilities, both in-furnace scrubbing and flue-gas scrubbing are planned. As will be detailed subsequently, testing by Gotaverken at an operating CFB facility in Sweden has demonstrated that highly-efficient control of both SO_2 and HCl can be achieved by limestone-injection scrubbing in the furnace, without flue gas scrubbing. The testing also determined, however, that high-efficiency HCl scrubbing in the furnace alone, while achievable, is not practical. SO_2 removal reactions prefer high (i.e. furnace) temperatures; however, HCl removal reactions prefer cooler temperature regimes. Gotaverken has determined that highly-efficient HCl removal in the furnace requires large amounts of limestone injection compared with what amounts of limestone are needed to efficiently scrub SO_2 in the furnace. Further, Gotaverken has found that injection of excessive amounts of limestone into the furnace to scrub HCl there leads to an undesired increase in NOx emissions. Accordingly, Gotaverken has decided to split the scrubbing duties in its design: SO_2 will be scrubbed efficiently in the high-temperature regime of the furnace by injection of moderate amounts of limestone, and HCl will be scrubbed efficiently in the cooler zone downstream of the furnace using a system in which hydrated lime is dry-injected into a ductwork venturi placed just ahead of the baghouse. The venturi serves to distribute the injected lime evenly over the fabric-filter surface, which is where most of the HCl removal then occurs.

ENVIRONMENTAL TEST DATA

Environmental test data are available for the WLSSD bubbling-bed facility, and also for Gotaverken CFB units in Scandinavia fueled with RDF.

WLSSD FACILITY, DULUTH, MINNESOTA

Emissions from one unit of the WLSSD facility were tested in 1987 by Interpoll Laboratories.(5) According to the test

TABLE 2: DESCRIPTIONS OF PLANNED ERIE AND ROBBINS CFB FACILITIES

Facility Name/ Location	Erie Resource Recovery Facility, Erie, PA	Robbins Resource Recovery Facility, Robbins, IL
Private Owner/Operator	Reading Energy Group	Reading Energy Group
Fuel	RDF (2" max.)	RDF (4" max.)
Furnace/Boiler Type	CFB Waterwall (Gotaverken)	CFB Waterwall (Gotaverken)
Combustor Capacity	Two units @ 355 TPD each	Two units @ 600 TPD each
Bed Temperature	1,500-1,700 F	1,500-1,700 F
Excess Air	Approximate 30%	Approximate 30%
Steam Production	110,000 lb/hr per unit @ 800 psig, 770 F	225,000 lb/hr per unit @ 835 psig, 775 F
Energy Recovery	23 MW electric power	47.5 MW electric power
Air Pollution Controls	In-furnace scrubbing (SO_2) Flue-gas dry-injection scrubbing (HCl) Baghouse	In-furnace scrubbing (SO_2) Flue-gas dry-injection scrubbing (HCl) Baghouse
Status	Permits granted 1987	Permit applications submitted

Information Source: Alternative Resources, Inc.

report, the unit, while tested, was firing about five tons per hour of RDF and about fourteen tons per hour of sewage sludge.

Emissions test results are summarized in Table 3. The venturi scrubber achieved very low particulate emission rates, on a par with modern fabric filters. The tray scrubber reduced HCl to a minuscule level, although SO_2 emissions appear several times higher than would likely have been the case with a conventional flue-gas scrubber. CO levels were low (less than 31 ppm) for 62 of 64 fifteen-minute periods of continuous CO monitoring during the test program. During two fifteen-minute periods, a combustion upset arising from a fuel-feed problem resulted in CO levels in excess of 300 ppm.

The tested dioxin levels are on a par with the lowest achieved by any technology to-date in the United States. The low dioxin emissions demonstrate the high combustion efficiency achieved by fluid-bed combustors at their inherent operating temperatures near 1,500°F, and likely also reflect additional dioxin removal efficiency through condensation processes in the wet scrubbers.

The tested NOx emissions illustrate the inherent "low-NOx" features of fluid-bed combustion, and may also reflect additional NOx removal by the wet scrubber systems.

GOTAVERKEN CFB FACILITIES IN SCANDINAVIA

The unique environmental design features claimed for CFB combustors include low NOx emissions; high combustion efficiency,with resultant low dioxin emissions; and efficient scrubbing in-furnace. Gotaverken has supplied emissions test data for their CFB plants operating in Scandinavia. Data are presented in Table 4 for the Sande Plant in Norway, tested in 1986 while combusting 100 percent RDF. When tested, the Sande Plant had an ESP for control of particulate emissions, and the plant employed neither in-furnace scrubbing nor a flue-gas scrubber.

Data are presented in Tables 5 and 6 for the 200 TPD Sundsvall Plant in Sweden, tested first in 1985 while burning RDF in combination with other fuels, and tested again in 1988 while burning 100 percent RDF.

In 1985, the Sundsvall Plant was equipped with an ESP, but neither in-furnace scrubbing, nor flue-gas scrubbing were

TABLE 3: TESTED EMISSIONS FROM THE WLSSD FACILITY

Test Date	November 1987
Fuel	RDF (26%), Sewage Sludge (74%)
Combustion Temperature	Approximately 1,475°F in bed
Air Pollution Controls	Cyclone, venturi wet scrubber, wet tray-type scrubber
Tested Emissions	
o Particulate	.004-.006 gr/dscf @ 12% CO_2
o SO_2	92-121 ppmv, wet[a]
o HCl	< 1 ppmv, wet[a]
o CO	1-31 ppmdv @ 7% O_2
o NOx	15-43 ppmdv @ 7% O_2
o Dioxin (EPA Toxic Equiv.)	.09-.14 ng/Nm^3 @ 7% O_2

[a] Concentrations if corrected to a "dry basis @ 7% O_2" would be within ± 10% of values reported here.

TABLE 4: EMISSION TEST RESULTS FOR GOTAVERKEN CFB BOILER AT SANDE PAPER MILL, NORWAY

	Test A (1986)	Test B (1986)	Test C (1986)
Fuel	100% RDF	100% RDF	100% RDF
Combustion Temperature (^{o}F)	1450	1500	1510
Air Pollution Controls	ESP; no in-furnace or flue-gas scrubbing		
Emissions			
CO (ppmv)	89	96	73
NO_x (ppmv)	129	110	168
Dioxin Test Equiv. Eadon Method (ng/Nm^3)	0.7	1.4	0.5

Source: Gotaverken Energy Systems, Inc.

TABLE 5: 1985 EMISSION TEST RESULTS FOR GOTAVERKEN CFB FACILITY AT SUNDSVALL, SWEDEN

	Test A (4/12/85)	Test B (4/12/85)	Test C (9/27/85)	Test D (9/27/85)
Fuel Mix	RDF/Peat	RDF/Peat	RDF/Tires/ Wood Chips	RDF/Tires/ Wood Chips
Combustion Temperature (°F)	1562	1508	1598	1598
Air Pollution Controls	------ESP; no in-furnace or flue-gas scrubbing-------			
Emissions				
CO (ppmv)	30	33	95	20
Dioxin Toxic Equiv. Eadon Method (ng/Nm^3)	0.4	1.3	0.6	1.8

Source: Gotaverken Energy Systems, Inc.

TABLE 4: 1988 EMISSION TEST RESULTS FOR GOTAVERKEN CFB FACILITY
AT SUNDSVALL, SWEDEN

	Test Number					
	1 (2/23/88)	2 (2/24/88)	3 (2/24/88)	4 (3/2/88)	5 (3/2/88)	6 (5/19/88)
Fuel	----------100% RDF----------					
Combustion Temperature	----------Approximate 1,500°F----------					
Air Pollution Controls	BH	BH	IFS/BH	IFS/BH	IFS/BH	IFS/BH
	(Baghouse only)		(In-furnace scrubbing, plus baghouse)			
Furnace Limestone Injection (kg/hr)	0	0	500	500	500	500
Emissions (ppmdv @ 7% O_2)						
CO	9	9	18	4	5	13
NOx	152	163	333	337	333	277
Dioxin Toxic Equiv., Eadon Method	6.2	7.8	0.7	0.7	1.0	1.4
	(w/o scrubbing)		(------------with scrubbing------------)			
HCl	285	324	19	13	14	10
	(uncontrolled)		(93-97% reduction from uncontrolled levels)			
SO_2	55	53	<3	<3	<3	<3
	(uncontrolled)		(--95% reduction from uncontrolled levels--)			

Notes:

a Sundsvall is a 200-TPD CFB facility operated at 95% of full load during all tests.

b Sampling and analysis by Swedish test methods, similar to EPA-approved methods commonly used on incinerators in the United States.

Data Source: Gotaverken Energy Systems, Inc.

utilized. In 1988, the ESP had been replaced with a baghouse, and testing was performed both with and without in-furnace scrubbing. The Sundsvall Plant, however, does not have a flue-gas scrubber.

The tested NOx emissions were 100-168 ppm at the Sande Plant. Testing at the Sundsvall Plant in 1988 showed 152-163 ppm NOx with no limestone injection into the furnace, but over 300 ppm with injection of large amounts of limestone (500 kg/hr for a 200-TPD RDF combustor). Recall from an earlier discussion that Gotaverken's current design restricts the amount of limestone injection into the furnace, to ensure low NOx emissions. With this design, Gotaverken guarantees NOx emissions to be less than 200 ppm, or about two-thirds or less of the emission rate guaranteed typically for conventional combustors.

The test data for the Sande and Sundsvall Plants provide evidence that RDF-fired CFB combustors are capable of high combustion efficiency at their inherent operating temperatures of near 1,500°F. At the Sande Plant, CO levels of 73-96 ppm were measured. At the Sundsvall Plant in 1985, tested CO levels ranged from 30-95 ppm. On this basis, it had been thought that CO emission levels for fluid-bed combustors, while low, might be about twice the levels typically achieved by modern, conventional combustors (20-50 ppm). However, the 1988 CO test data from Sundsvall (4-18 ppm), and the Duluth CO test data presented earlier indicate fluid-bed combustors can now achieve the same, low CO emission levels as modern, conventional combustors. Experience at both the Sundsvall and Duluth Plants, however, reveals that fuel feed to the combustor must be managed carefully to avoid combustion "upsets" and attendant CO excursions.

Tested dioxin emissions for the ESP-equipped Sande Plant and Sundsvall Plant (1985 tests) were similar and in the range of 0.4 to 1.4 ng/Nm^3 @ 7% O_2 of 2,3,7,8 TCDD toxic equivalents by the Eadon method. Testing at the baghouse-equipped Sundsvall Plant (1988 tests) showed similar dioxin emission levels for all tests where efficient HCL removal was achieved through in-furnace scrubbing. However, dioxin tested higher at the Sundsvall Plant by an order of magnitude during two tests when no in-furnace scrubbing was carried out. Based on the test data, Gotaverken expects that the Erie and Robbins CFB facilities, which will employ in-furnace (and flue-gas) scrubbing, will not exceed a dioxin toxic-equivalent emissions rate of 2 ng/Nm^3 @ 7% O_2.

In Table 7, the tested dioxin emissions from the Duluth and Sundsvall fluid-bed facilities are compared with dioxin emissions from modern, conventional combustors tested in the United States since 1986. Dioxin emissions from the fluid-bed facilities are seen to be consistent with the range of tested emissions for the most-modern conventional facilities in the United States.

The effectiveness of in-furnace scrubbing in Gotaverken CFB combustors fueled with RDF is demonstrated in Table 6. By comparing the SO_2 and HCl results for the test runs with and without in-furnace limestone injection, (Tests 1 and 2, versus Tests 3-6) the effectiveness of in-furnace scrubbing is evident. Both SO_2 and HCl removal were approximately 95 percent with in-furnace scrubbing. The test data indicate that efficient scrubbing of both HCl and SO_2 can be achieved with in-furnace scrubbing sufficient to meet typical permit limits (70-75% SO_2 removal, 90% HCl removal). However, again, as was noted earlier, in order to scrub HCl with high efficiency in the furnace, a high limestone injection rate is needed, which in turn, as illustrated by the data in Table 6, leads to NOx emissions greater than desired. At the planned Erie, Pennsylvania and Robbins, Illinois facilities, therefore, SO_2 will be scrubbed efficiently in the CFB furnaces using about one-quarter the limestone injection rate per ton of RDF as was used during the Sundsvall tests. At the United States facilities, HCl will be controlled with an all-dry flue-gas scrubbing system (hydrated lime injected into a ductwork venturi just prior to the baghouse). A similar all-dry flue-gas scrubber/baghouse system is used at the Wheelabrator 200-TPD mass-burn facility operating in Claremont, New Hampshire. Compliance testing at the plant in May 1987 showed this system to achieve 95 percent HCl removal, and 78-84 percent SO_2 removal.(10)

Based on the test data presented here, the SO_2 and HCl removal efficiencies required typically to meet permit limits (70 - 75% SO_2 removal, 90% HCl removal) can be met by the Robbins and Erie facilities with either the in-furnace or flue-gas scrubbing systems alone. However, by splitting the scrubbing duty between the in-furance system (SO_2) and flue-gas system (HCl), limestone use in the furnace is minimized, preserving the inherent "low-NOx" performance of the CFB furnace.

The emissions limits imposed on the Erie CFB facility by its 1987 permit are summarized in Table 8, as are the emissions limits proposed in 1988 for the planned Robbins CFB facility. Proponents of CFB waste-to-energy projects now

TABLE 7: MEASURED DIOXIN/FURAN EMISSION CONCENTRATIONS AT MODERN RESOURCE RECOVERY FACILITIES

Facility	Size/Type[a]	Air Pollution[b] Control Equipment	PCDD + PCDF Emission Concentration ng/Nm3 @ 7% O2 Eadon Toxic Equivalent
Fluidized-Bed Incinerators			
Duluth, MN	400/RDF	MC/WS	0.09 - 0.16[c]
Sundsvall, Sweden	200/RDF	IFS/FF	0.5 - 1.1 (4 tests)[d]
			Approx. Avg. = 0.5 (Eadon)
Grate-Burn Incinerators			
Commerce, CA(6)	300/MB	SDA/FF	0.03
Marion County, OR(7)	1100/MB	SDA/FF	0.2
Bristol, CT(8)	650/MB	SDA/FF	0.1[3]
Millbury, MA(9)	1500/MB	SDA/ESP	0.6
Claremont, NH(10)	200/MB	DS/FF	0.4[3] (Unit 1)
			0.5[3] (Unit 2)
Hartford, CT(11)	2025/RDF	SDA/FF	1.95[3]
Biddeford, ME(12,13)	607/RDF	SDA/FF	0.07
			Approx. Avg. = 0.7 (Eadon)

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a) Size represents total facility waste throughput in tons-per-day (TPD)
MB - mass burn RDF - refuse derived fuel

b) IFS - in-furnace scrubbing FF - fabric filter
MC - mechanical collector ESP - electrostatic precipitator
WS - wet scrubber SDA - spray dryer absorber
DS - dry scrubbing (dry in-duct injection)

c) EPA toxic equivalency factors (results in a concentration approximately one-half to two-thirds of the concentration using Eadon method)

d) 1988 tests at Sundsvall Plant conducted with in-furnace scrubbing.

TABLE 8: PERMIT LIMITS FOR PLANNED CFB FACILITIES

Pollutant/Parameter	Erie, PA Facility Existing Permit (1987)	Robbins, IL Facility Limits Proposed by Proponent (1988)
Particulate - Total	0.015 gr/dscf	0.010 gr/dscf
Particulate - PM10	No Limit	0.010 gr/dscf
Hydrochloric Acid	30 ppmdv (1 hour), or 90% control (1 hour)	50 ppmdv (1 hour) or 90% control
Sulfur Dioxide	50 ppmdv (1 hour), or 70% control (1 hour)	30 ppmdv (8 hour), or 75% control
Carbon Monoxide	100 ppmdv (8 hour), or 32 lb/hr	100 ppmdv (1 hour),
Combustion Efficiency	99% (1 hour)	99.9% (4 hour)
Dioxin/Furan	2.0 ng/Nm^3 toxic equivalents (EPA Method)	BACT
Nitrogen Oxide	No specific limit	200 ppmdv (30-day)
Opacity	<10%, except <30% for period(s) totalling not more than 3 minutes-per-hour	<10%, except <30% for period(s) totalling not more than 3 minutes-per-hour
Combustion Temperature	Aux. fuel burners engage if combustion temperature drops below 1650°F	Combustion temperature ≥ 1500°F, gas residence time ≥ 3 seconds

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a All concentration values (gr/dscf and ppmdv) are stated as corrected to 7% O_2.

Information Source: Alternative Resources, Inc.

appear confident in guaranteeing emissions levels on a par with the strictest performance guarantees made for conventional combustors in the United States.

ASH RESIDUE CHARACTERISTICS FOR CFB COMBUSTORS

With regard to ash residue, the EPA Extraction Procedure "EP" Toxicity test is used to determine whether a plant's residue is likely to cause toxic trace-metal leachate when the residue is disposed in a landfill. Results of such testing on residue from the Gotaverken RDF-fired CFB boiler at Sundsvall, Sweden are presented in Table 9.

Testing of residue leachate in the United States has generally indicated that bottom ash from conventional waste-to-energy facilities passes the EP Toxicity Test, and combined (blended) bottom and fly ash normally passes. Fly ash alone, however, from these facilities fails the EP Toxicity Test, because the test results indicate that the leachate contains unacceptably-high levels of lead, cadmium and sometimes other metals. What is striking about the Sundsvall residue data in Table 9 is that both the bottom ash and fly ash are shown to pass the EP Toxicity Test by large margins. The fact that fly ash from conventional facilities in the United States tests to be "toxic," and fly ash from the Gotaverken CFB facility in Sweden clearly does not, may result in part from differences in the composition of Swedish versus United States solid waste; however, this appears unlikely. Given that the two countries enjoy similar standards of living and have similar patterns of consumption, there should be no marked difference in waste composition. Gotaverken offers a plausible explanation for the non-toxicity of CFB fly ash. Trace metals, volatized from the waste during combustion, are chemically "fixed" at high temperature onto particles of ash, bed sand (silica) and limestone circulating, within the furnace. Because of long residence time available in the furnace for reaction to take place at high temperature, the metals bond tenaciously to the inert particles. Metals "fixed" at high temperature to inert particles strongly resist leaching. Additional leachate testing of residue from Gotaverken CFB plants will be needed to determine whether these initial test results are repeatable.

TABLE 9: EP TOXICITY TEST RESULTS FOR GOTAVERKEN CFB RESIDUE

	Allowable Limit (ppm)	Sundsvall CFB[a]	
		Bottom Ash	Fly Ash
Arsenic	5.0	.007	.17
Barium	100.0	.1	.3
Cadmium	1.0	.05	.03
Chromium	5.0	.12	.04
Lead	5.0	.35	.04
Mercury	0.2	.0002	.001
Selenium	1.0	.01	.02
Silver	5.0	.04	.02

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[a] Sundsvall CFB facility burning RDF, with ESP and with in-furnace limestone scrubbing. Bottom and fly-ash samples obtained 4/17/86 and analyzed by an EPA-certified, B.A.W. Laboratories, Inc., of North Carolina.

SUMMARY

Although experience with fluidized-bed technology in the United States is limited for waste-to-energy applications, interest in this innovative technology is rapidly increasing, owning to the operational and environmental advantages claimed for it. Attributes shared by both bubbling fluidized-bed (BFB) combustors and circulating fluidized-bed (CFB) combustors include low excess air requirements; highly efficient, stable combustion; and low NOx emissions. In addition, CFB combustors provide multi-fuel flexibility and highly-efficient (in-situ) scrubbing of SO_2.

Environmental test data from Duluth, Minnesota and Scandinavia indicate that RDF-fueled BFB and CFB combustors both achieve NOx emissions roughly two-thirds or less than the levels associated with conventional combustion technologies for solid waste. These test data also demonstrate that fluid-bed combustors, with their operating temperatures inherently near 1,500°F, have low dioxin emissions consistent with the emissions from the most modern of conventional combustors in the United States. Additional test data from Sweden are available for the CFB design that demonstrate SO_2 removal efficiencies with in situ scrubbing technology comparable to removal efficiencies achieved with flue-gas scrubbers.

Fly ash from conventional mass-burn and RDF-fired facilities typically fail the EPA Extraction Procedure "EP" Toxicity Test for trace metals. Fly ash from the Gotaverken CFB facility has been tested and passes the EPA toxicity test by a substantial margin. Landfill disposal of the fly ash from the CFB boiler, therefore, could be less controversial, if additional testing confirms initial findings of non-toxicity.

NOTICE

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necesarily reflect the views of the Agency and no official endorsement should be inferred.

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DISPOSAL OF DOMESTIC REFUSE AND INDUSTRIAL REFUSE IN SWEDEN 1987

ROGER BERGSTRÖM
STOCKHOLM ENERGI

WASTE MANAGEMENT IN SWEDEN

The Kingdom of Sweden has a population of about 8,4 million on an area of about 450 000 km^2.

POLICY

The Swedish Government has during 1986-87 given the following guidelines for the future solid waste management in Sweden.

1. The quantity of waste should be reduced by measures taken on boath the production side and the consumptionside. This should be a basic and common community task. Also the producers have a responsibility for the protection of the environment.

2. Produced waste should be reused or recovered as far as it is practically and economically possible.

3. Waste that must be processed should be handled and disposed of in such a way that negative effects on the environment will be minimized in both the short and long term. Stringent environmental protection requirements must thus be estabished.

4. The handling of hazardous waste must be improved. Substances and products which, after being mixed with other types of waste, cause special damage to the environment should be handled separately.

5. There must not be any one-sided emphasis on a particular method of waste treatment, or a situation which forces one particular method to be adopted. Alternative methods should be developed, tested and tried.

In 1985 the Swedish Association of Public Cleansing an Solid Waste Management presented its wiews on waste management in the future in a report titled "Future Waste Treatment in Sweden - A Combination of Methods to Conserve Resources and Improve the Environment". In summary the following was stated:

The management of the Swedish waste mountain must be based, both now and in the future, on a combination of different methods, where the choice of method must put environmental protection first. There must not be any contradiction between the different methods; instead they should be regarded as complementing each other. Only in this way the waste problem can be solved in an optimal fashion. Large scale and small-scale methods must go hand in hand and complement each other.

In the future it would then appear that emphasis within solid waste management should be placed on the following:

* Recovery of material in households and industries, involving direct action by residents and industries in cooperation with those responsible for waste collection.

* Thermal treatment of waste with energy recovery (provided that the environmental consequenses of emissions can be kept under control even in the future).

* Landfilling, which always will be needed regardless of other methods utilized, for non-reclaimble materials, residual products from other treatment etc.

* Continued reaserch and development on central mechanical separation and composting of waste.

* Development of large-scale methodes for production of biogas out of organic waste.

* Increased utilization of material and energy resources in industrial waste. The conservation of resources should chiefly take place in the industries themselves, by using low-waste technology and by internal recovery of the waste products.

* Improved collection and treatment of hazardous wastes from industry.

The representatives of waste management in Sweden are trying to minimize the environmental effect of produced waste by means of far-reaching environmentprotective measures and in order not to cause further damage. The causes of environmental disturbances are usually to be found far back in the chain - in production and consumption.

AMOUNTS

There is a yearly production of household waste in Sweden of approx. 2,5 Mtons, that means approx. 300 kg/person and year or approx. 750 kg/household and year. Depending upon social conditions, livingstructure, geographical location the amount/person and year varies between 150-450 kg. The total amount of houshold waste is not supposed to increase. On the contrary it is supposed to decrease due to an increased recovery of different materials out of waste.

The yearly production of industrial waste brought to final disposal and treatment in the municipalities is estimated to about 4,5 Mtons. Due to the varying industrialstructure from municipality to municipality the amount of industrial waste also varies to a great extent.

The total amount of hazardous waste is most uncertain, but is estimated to about 0,5 Mtons.

Waste treatment and disposal in Sweden 1987 (after source recovery; % of weight):

Category	Land-application	Incineration	Separation compost	Land-filling	Total Mton
Household waste		55 %	10 %	35 %	2,5
Industrial waste		5 %		95 %	4,5
Sewage sludge	60 %			40 %	0,4

Around 50 % of the industrial waste is estimated to be combustible. This fraction (pallets, cable drums, demolition timber etc) is normaly sorted out at dumping site. By offering better service and lowercharges, industrial companies are encouraged to separate the waste at source into different containers.

Since 1987 all swedish municipalities are taking part in a campaign to collect small hazardous batteries. The information includes advertisements in newspapers, on TV, on buses etc. It also includes a special information to all schools. The households can deliver the batteries in special battery-boxes (located on streets, markets etc), to shops where batteries are sold and sometimes in connection with the collection of ordinary household waste. At present, about 50 % of the mercury content in sold batteries is returned.

In 1982, the Parlament made a decision on the recovery of aluminium cans used for beverages, whereby at least 75 % was to be recovered not later than 1985. A special company was formed to run a deposit system. The deposit system was introduced in 1984. During 1985 66 % of the cans were recovered amount during 1986 was approx the same. To achieve the recovery level of 75 % the consumer$ economic incentive was doubled during 1987, to SEK 0,50 per can. For the moment about 85 % of the cans are returned.

Also for glass bottles for bevarages a special deposit system is used. Each bottle is in average returned and reused 33 times.

Waste recovery has also reached significant levels with regard to certain types of industrial materials. The system that have been developed within both municipalities and industries to recover residuals and to recycle materials have eliminated any increase in the amount of waste generated during the last ten years.

REFUSE INCINERATION IN STOCKHOLM

Large-scale refuse incineration has been employed in Stockholm since way back in 1906. The furnaces then had a total daily capacity of 160 tons.

Since those early days, refuse incineration has developed dramitically from an essential but primitive method used for refuse disposal to an efficient refuse-fired process with a high degree of energy recovery and advanced flue-gas cleaning.

COMPOSITION

Domestic refuse in Stockholm city amounted to around 220 000 tons in 1988. During the past 5 years, the annual quantity has remained largely unchanged, possibly with a moderate upward trend.

In addition to domestic refuse, substantial quantities of industrial refuse are generated in Stockholm. But since the disposal of this type of refuse is not monopolized by Stockholm city authorities, no reliable statistics are available on the quantities and composition.

The annual quantity of industrial refuse generated in Stockholm is estimated to be around 200 000 tons. Half of this total, or around 100 000 tons, is considered to be suitable for incineration combined with energy generation. But taking into account the practical problems, control of refuse flows, etc, the actual quantity available as fuel is estimated to be about 50 000 tons.

Paper, cardboard, plastic film and wood are sorted out of the selected industrial refuse, and these serve as the raw material for the production "crushed fuel". The raw material passes through a crusher, followed by magentic separation.

In addition, wood-chip fuel is produced from demolition timber, construction site waste, pallets and similar wood waste which is sorted at source.

Characteristic particulars of the various refuse-based fuels are tabulated below.

		Domestic refuse	Crushed fuel	Wood-chip fuel
Lower heating value (LHV)	MJ/kg	11	13-14	15
Moisture	%	33	10-20	15-20
Ash	%	21	5-10	5-7
Paper	%	39	50-70	
Plastics	%	9	3-10	
Food residues	%	30		
Metals	%	3		
Glass	%	7		
Wood	%	1	10-20	95
Garden waste	%	4		
Textiles	%	3	1-6	
Remainder	%	4	5-10	

REFUSE-FIRED COMBINED HEAT AND POWER PLANT IN STOCKHOLM

In order to relieve an older domestic refuse incineration plant in nothern Stockholm, a new plant - now known as the Högdalen plant - was built in 1969.

In the initial stages, the plant incinerated annually about 125 000 tons of Stockholms domestic refuse. The boilers were in continuous operation for 5 days a week, and they were shut down during the weekends.

The plant consisted of two refuse-fired steam boilers, each with a capacity of 11 tons of refuse per hour, corresponding to a thermal rating of around 27 MW per boiler. The boilers generated superheated steam at 36 bar and 360°C. A steam system common to the two boilers supplied steam to a condensing trubine with an electrical rating of 24 MW. The steam was condensed without heat recovery in an air-cooled condenser.

The plant was planned from the outset for two additional refuse-fired boilers of the same rating to be installed at a later date. The preparations concerned matters such as the refuse bunker, boilerhouse, slag bunker and the turbine rating.

The plant was modified in 1978 and 1979 to enable it to supply heat to an adjacent district heating system and thus ensure better utilization of the heat content of the refuse. The turbine was conwerted to back-pressure operation, which involved removing the last turbine stages and raising the exhaust pressure from 0.05 bar to 0.6 - 1.1 bar.

The inlet stages of the turbine were also modified, which enabled the electrical rating of the turbo-generator to be maintained. In addition, an 80 MW oil-fired steam boiler and two oil-fired hot water boilers of 10 MW each were installed for stand-by and peak load duties. A hot water accumulator with a volume of 900 m^3 was installed to ensure that the deliveries of heat to the district heating system would not be affected by fluctuations in the steam flow caused by variations in the heating value of the refuse.

At the same time, the plant was put on continuous operation, which increased the annual refuse incineration capacity to 155 000 tons. This raised the utilization time of the boilers to 7 000 h/year.

Due to an increase in the heat demand, an electric boiler was installed in 1982 for generating heat for the district heating system.

After 15 years of operation, the refuse incineration boilers began to show signs of wear, with recurrent tube leakage in the superheater and convection sections. A new boiler was therefore obviously necessary to avoid the need for periodic reductions in the incineration capacity and to provide scope for rational maintenance. Moreover, an increase in the boiler capacity would enable all domestic refuse from Stockholm city to be incinerated in the Högdalen Plant, and the earlier incineration plant could then be shut down. The installed boiler capacity would also allow other solid fuels, such as wood chips and processed industrial refuse, to be fired during the winter to replace fuel oil.

The decision to install a third refuse-fired boiler was taken in the autumn of 1983. The order for the boiler was placed in the spring of 1984, and the boiler was taken into commercial operation in the beginning of January 1986.

In conjunction with the modernization of the Högdalen Plant, the two earlier boilers were subjected to extensive technical refurbishment in cooperation with the boiler supplier.

NEW REFUSE INCINERATION BOILER (P3)

The boiler was installed in the place prepared for it in the existing boilerhouse. The size of the place reserved for it allowed for a boiler with a rating of 15 tons of refuse per hour, which corresponds to a steam generation capacity of about 50 tons per hour.

The boiler erection work was rendered difficult by the fact that it had to be carried out in an existing building and between two existing boilers. In spite of this, the erection time was very short.

The order for the plant was placed in week 25 of 1984, erection of the boiler was started in week 49 of the same year, and the boiler was ready for firing in week 48 of 1985. After approval of the performances tests, the boiler was taken over by Stockholm Energy in March 1986.

The incineration furnace and boiler were purchased from Martin Gmbh of Munich, West Germany, with Wehrle Werk AG of Emmendingen, West Germany, and Ångpanneservice of Norrköping, Sweden acting as sub-contractors.

Our procurement was based on the principle that the lowest price would not be decisive to the choise of supplier. Investment in a refuse-fired power generation boiler must be assessed on the basis of an overall assessment, in which availability, performance, operating costs and maintenance costs are weighed into the cost of the investment itself.

Matters to which we assigned particular importance included:

- availability
- operation time between overhauls
- combustion efficiency
- service life of superheater tubes
- performance
- maintenance requirements and costs
- robust and safe design

The design of the boiler is shown in Figure 1.

The boiler consists of three empty vertical uptakes, followed by the superheater, evaporative surfaces and economizer mounted in a horizontal section. Rapping gear is used for on-load cleaning of all surfaces in the horizontal section of the boiler. No sootblowing system is installed.

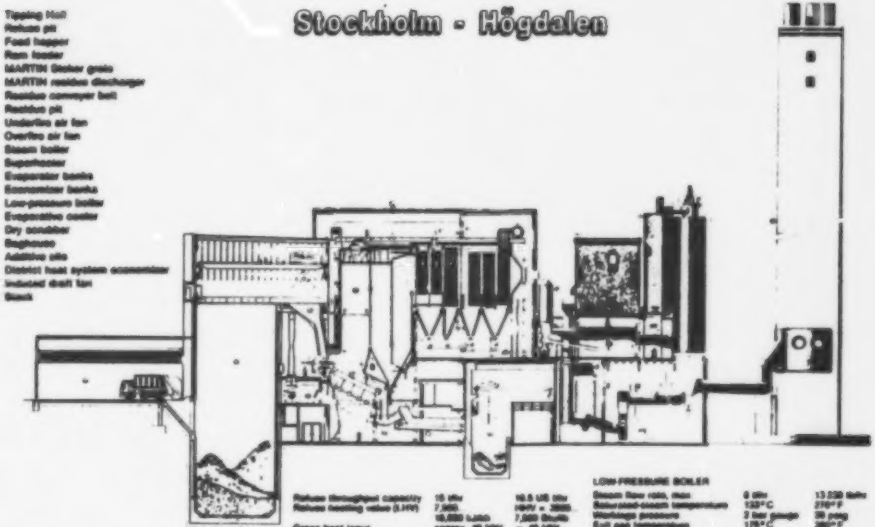
The stoker consists of three parallel travelling grates, each divided into 5 zones with separate supply of primary air under the stoker grate. The air flow rate to each zone can be controlled separately, and the flow rate to each of the 15 inlet points (3 grates with 5 zones each) can be measured individually.

The refuse incinerated in the Högdalen Plant has a heating value of about 11 MJ/kg. Due to the high heating value, a furnace temperature of 800°C can always be maintained. It is estimated that the dwell time of the flue gases at 800°C is around 2 seconds at full load.

In order to minimaze the risk of possible future tube damage, the boiler was designed for low flue gas velocity and low temperature upstream of the superheaters. The flue gas velocity is below 5 m/s at all points in the boiler. At full load and with the boiler fouled, the temperature upstream of the superheaters is below 660°C.

Stockholm - Högdalen

1 Tipping Hall
2 Refuse pit
3 Feed hopper
4 Ram feeder
5 MARTIN Stoker grate
6 MARTIN residue discharger
7 Residue conveyor belt
8 Residue pit
9 Underfire air fan
10 Overfire air fan
11 Steam boiler
12 Superheater
13 Evaporator banks
14 Economizer banks
15 Low-pressure boiler
16 Evaporative cooler
17 Dry scrubber
18 Baghouse
19 Additive silo
20 District heat system economizer
21 Induced draft fan
22 Stack



Refuse throughput capacity	15 t/hr	16.5 US t/hr
Refuse heating value (LHV)	7,950- 18,050 kJ/kg	HHV = 3800- 7,800 Btu/lb
Gross heat input	approx. 40 MW	≈ 40 MW
Steam flow rate, max	approx. 50 t/hr	110 230 lb/hr
Live steam pressure (S. H. outlet)	36 bar	520 psig
Live steam temperature	350°C	660°F
Feedwater temperature	133°C	270°F
Exit gas temperature	260°C	500°F
Turboalternator o[illegible]	24 MW	≈ 24 MW

LOW-PRESSURE BOILER

Steam flow rate, max	6 t/hr	13 230 lb/hr
Saturated-steam temperature	133°C	270°F
Working pressure	2 bar gauge	29 psig
Exit gas temperature	175°C	350°F

DISTRICT HEAT SYSTEM ECONOMIZER

Water flow rate	56.8 t/hr	125 230 lb/hr
Water temperature	in. 70°C	160°F
	out. 120°C	250°F
Flue gas temperature	in. 165°C	330°F
	out. 85°C	185°F

The dwell time of the flue gases up to the first superheaters is estimated to be about 12 seconds at full load. No. 1 superheater is located first in the horizontal section of the boiler and the tubes are arranged in the direction of the flue gas flow, in order to ensure the lowest possible material temperature.

With the heat transfer surfaces fouled, the flue gas temperature after of the economizer, the temperature is reduced further to around 190°C in a low-pressure boiler. After the low-pressure boiler, the flue gases are cooled to around 140°C in a feed-water cooled boiler and are then disharged to the flue gas cleaning plant.

Operating conditions of boiler P3

Fuel consumption	15 ton/h
Lower heating value (refuse)	11 MJ/kg
Air flow: primary	55 000 m^3/h
secondary	20 000 m^3/h
O_2 content after boiler	7.5 % of humid gas
CO content	25-40 ppm
High-pressure boiler	35 bar (abs) at 360°C
	52 t/h, 37 MW
Heat recovery boilers LP steam	2.5 bar (abs) at 130°C
	1.8 MW
Feed water	145°C
	1.8 MW
District heating	110°C
	1.0 MW
Flue gas temperature in stack	110°C
Overall efficiency	85 %

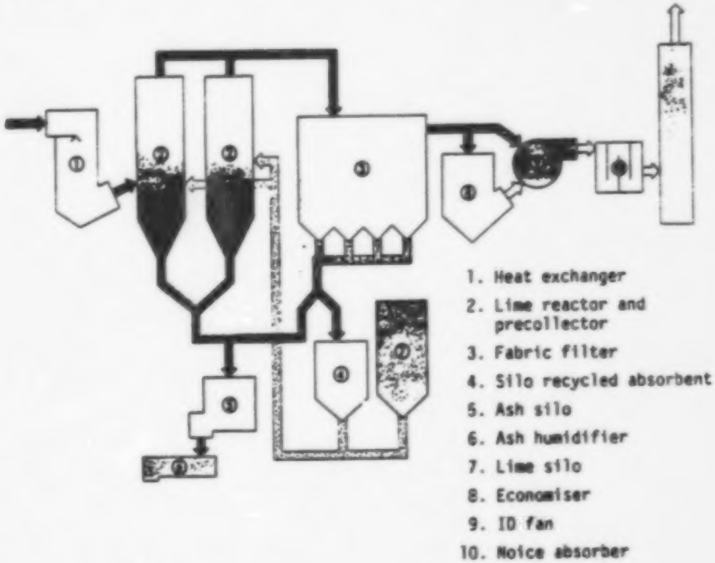
NEW FLUE GAS CLEANING EQUIPMENT

Flue gas cleaning equipment consisting of lime reactors, baghouses and flue gas fan were supplied by fläkt Industri AB .

The guiding selection criteria were as follows:

- environmental performance
- technology
- high availability
- economy
- potential for adapting to future demands

The arrangement of the flue gas cleaning equipment is shown diagramatically in Figure 2.


1. Heat exchanger
2. Lime reactor and precollector
3. Fabric filter
4. Silo recycled absorbent
5. Ash silo
6. Ash humidifier
7. Lime silo
8. Economiser
9. ID fan
10. Noice absorber

After the heat recovery boiler, the flue gases flow through two reactors, each of which is rated for half the flue gas flow. An Na_2S solution, possibly water for attemperation and $Ca(OH)_2$ (slaked lime) for neutralizing the acidic components are added to the flue gases in the reactors.

The flue gases then flow through a baghouse (fabric filter) which collects dust, products gererated in the reaction of the lime wit: acidic components and excess lime.

After the bag filter, the flue gases flow through an economizer cooled by district heating water, where they are cooled to about 110°C, and the flue gas fan then discharges the flue gases through the sound attenuator to the stack and out to atmosphere.

The mixture of dust, reaction products and axcess lime collected in the baghouse is discharged either to the dust silo or to the recycled dust silo. The filter dust is humidified before discharge into a truck for transport to the disposal site.

The dust from the recycled dust silo is mixed with lime from the lime silo and the mixture is metered into the reactors. Mixing of dust into the lime enables excess lime in the dust to be utilized, which reduces the total lime consumption.

The incineration of 220 000 tons of refuse generates about 46 000 tons of slag and around 9 000 tons of filter dust.

Operating conditions for flue gas cleaning equipment on boiler P3

Lime consumption	150 kg/h (at 20 mg HCl/m^3 at s.t.p)
Flue gas temperature in the filter	135°C
Filter cleaning	3-5 pulses/h
Sodium sulphide metering rate	2 kg/h

OPERATING EXPERIENCE

At the end of 1988, the boiler and flue gas cleaning equipment had been in operation for about 22 000 hours since commissioning, and had demonstrated high availability, apart from a few minor disturbances of an adjustment nature.

The energy generated in the three refuse incineration boilers represents a substantial proportion of the energy flow in the Högdalen network, as illustrated by the figures for 1988 tabulated below.

Heat distribution

Total	436 GWh	
From solid fuel	401 GWh	(87 %)
From oil	44 GWh	(9 %)
From electric power	18 GWh	(4 %)

Electric power generated

From refuse	48 GWh

The availability (excluding scheduled shut-downs) of boiler P3 and its flue gas cleaning equipment during 1988 was 99 %.

Our philosophy is to carry out mechanical cleaning and maintenance work during 2 to 3 weeks in the summer, when the heat demand is a minimum. The boiler is then operated continuously up to September, when it is shut down for 1 - 2 days for checking that the plant is in good condition before the winter season.

The boiler is then operated continuously up to April, when it is again shut down for 1 - 2 days for checking the plant and planning any repairs that may by needed during the next summer shut-down. No mechanical cleaning of the boiler is carried out during the short shut-downs mentioned above, and the rapping system is sufficiently effective to keep the boiler clean up to the summer shut-down. The boiler must thus be kept in operation for 8 000 hours between overhauls.

MOST RESTRICTIVE EMISSION REQUIREMENTS IN THE WORLD ?

The Government has specified the following environmental conditions for the Högdalen Plant.

Parameter		Emission requirement
Dust	(mg/m^3 of dry gas at s.t.p 10 % CO_2)	20
HCl	(mg/m^3 of dry gas at s.t.p 10 % CO_2)	100
Hg	(mg/m^3 of dry gas at s.t.p 10 % CO_2)	0,03
TCDD*	(ng/m^3 of dry gas at s.t.p 10 % CO_2)	0,1

*Dioxins measured as TCDD equivalents according to Eadon.

The normal raw composition is tabulated below.

All figures are applicable per m^3 of dry gas at s.t.p and 10 % CO_2.

Dust	1 000 - 5 000	mg/m^3 at s.t.p
HCl	500 - 1 000	mg/m^3 at s.t.p
SO_2	100 - 300	mg/m^3 at s.t.p
NO_x	300 - 450	mg/m^3 at s.t.p
Hg	100 - 400	µg/m^3 at s.t.p
TCDD	1 - 10	ng/m^3 at s.t.p

Numerous environmental measurements were carried out during 1986 - 1988, and the readings obtained are summarized below.

DUST: All measurements showed that the dust content of the clean gas was between 1 and 3 mg/m^3 at s.t.p.

HYDROCHLORIC ACID

Test	Raw gas mg/m^3 at s.t.p	Clean gas mg/m^3 at s.t.p	Temp °C	Lime kg/h
1986/1	-	51	173	
2	-	38	158	
3	-	-	-	-
4	1 100	63	168	180
5	729	85	175	138
6	852	28	173	243
7	814	46	150	97
8	795	32	150	137
1987/1	-	66	168	220
2	-	76	168	142
3	-	97	168	150
4	-	63	169	174
1988/1	885	63	169	190
2	874	50	163	160
3	822	64	155	140
4	583	21	144	160

SULPHUR DIOXIDE

Datum	Raw gas mg/m³ at s.t.p	Clean gas mg/m³ at s.t.p
1986/4	-	105
5	-	103
6	-	79
7	-	67
8	-	40
1987/4	-	100
1988/1	130	48
2	226	69
3	136	57
4	87	23

MERCURY

Datum	Raw gas mg/m^3 at s.t.p	Clean gas mg/m^3 at s.t.p
1986/1	-	85
2	-	150
3	-	-
4	-	25
5	414	51
6	308	33
7	-	46
8	-	58
1987/1	111	93
2	185	47
3	327	26
4	-	124
1988/1	238	122
2	330	20
3	108	16
4	91	12

DIOXIN
TCDD-equivalents

	Raw gas ng/m^3 at s.t.p	Clean gas ng/m^3 at s.t.p
1986	7.8	0.09
1987	4.0	0.1

During the autumn of 1988, flue gas cleaning equipment similar to that of boiler P3 was installed for the two earlier incineration boilers.

Decisive differences are that the plant is designed for 125°C in the reactors and that the dust discharge from the baghouse hoppers is designed for dust which is more difficult to handle.

The supplier can thus guarantee better collecting efficiency for Hg, SO_2 and CHl.

Performance tests are scheduled for April 1989, but preliminary measruements indicate excellent collecting efficiency.

Dust	2 mg/m³ at s.t.p
HCl	10 mg/m³ at s.t.p
SO_2	30 mg/m³ at s.t.p
Hg	5 µg/m³ at s.t.p

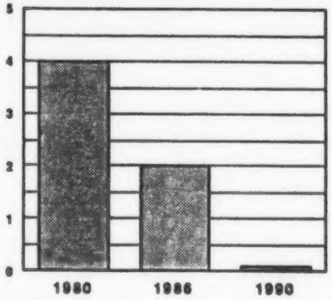
Normal operating conditions for all three incineration boilers as from 1989 will be:

Dust	1 - 3	mg/m³ at s.t.p
HCl	20	mg/m³ at s.t.p
SO_2	50	mg/m³ at s.t.p
Hg	30	µg/m³ at s.t.p
TCDD	0.1	ng/m³ at s.t.p

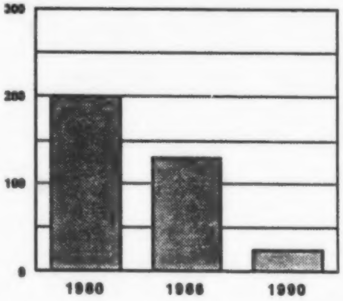
The bar cart below illustrates that the emissions of environmentally harmful substances has been reduced dramatically.

Continued environment conservation work in Stockholm will be oriented towards optimization of combustion, fine adjustment of the flue gas cleaning equipment and tstes on a pilot plant designed to reduce nitrogen oxides.

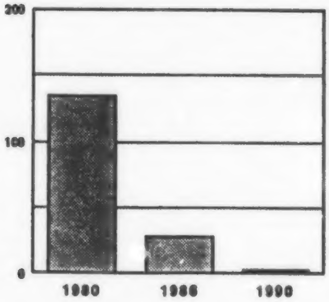
Dioxines, gram per year



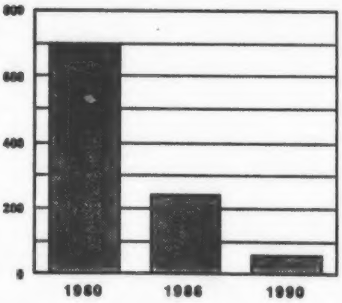
Hg, kg per year



Dust, ton per year



HCl, ton per year



RELATIONSHIPS BETWEEN VARIABILITY OF EMISSIONS AND RESIDUES FROM COMBUSTION OF MUNICIPAL SOLID WASTE AND WASTE COMPOSITION AND COMBUSTION PARAMETERS

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ABSTRACT

Emissions from combustion of MSW as well as the nature of flyash and bottom ash residues, are affected by the composition of the waste, conditions of combustion, furnace and boiler configuration, and the emission control system. Data from extensive diagnostic testing and data analysis by statistical and rational methods reveal the relationships between parameters and settings which can be used to determine the influence of operational factors to find optimum adjustments which will minimize dioxin and other emissions and undesirable properties of ash residues. Control systems employing oxygen and temperature measurements can maintain optimum combustion and minimum emissions. Continuous measurements of carbon monoxide can be used to evaluate the quality of combustion. Likewise, analysis of ash quality data can be used to evaluate the effectiveness of processing and mixing ash residues to minimize the leaching potential of heavy metals.

INTRODUCTION

The quantity and nature of gaseous and particulate emissions as well as composition and leaching characteristics of ash residues resulting from the combustion of municipal solid waste (MSW) is highly variable and dependent on many factors.

Waste composition varies continually, and moisture, especially, varies seasonally due to changes in ambient humidity and the presence of grass and leaves.

Particulate produced from combustion of municipal solid waste (MSW) depends on combustion efficiency and carbon burnout, combustion temperatures, lifting velocities, suspension versus bed firing, mixing of gases and underfire and overfire air distribution.

Inorganic emissions, especially metals, are associated with the particulate, but their quantity and chemical form are influenced by combustion temperatures and reactions with acid components of the waste.

Organic emissions depend on factors such as waste composition, flame temperature, excess oxygen, moisture, furnace configuration and volume (retention time).

The nature of the control system which varies feed rate, stoker action and air quantity and distribution also influences the stability of combustion and response to changes in feed composition, especially moisture content.

Due to the high variability resulting from these many factors, it is difficult to discern the influence of different parameters which affect the quality of combustion controls and which also the nature and even the leachability of flyash and bottom ash residues.

In spite of this variability, data from operation of MSW combustion facilities often exhibits surprising consistency. Variations in waste composition as well as in emissions from combustion are usually probabilistic in nature. They follow normal or log normal distributions that are confined within relatively narrow ranges that are more consistent and predictable than might be expected. The log normal distribution of data reflects random variations with roughly equal probabilities of occurrence. This property of the waste as well as of stack emissions and ash residues, facilitates the systematizing of data from individual plants and groups of similar technologies, making it possible to predict likely average and probable maximum emission rates of planned facilities, based on data from existing plants.(1)

While emissions seem to follow random probabilities of occurrence, influenced by seasonal variations in the nature of the waste, operating conditions also have a direct and independent bearing on the magnitude and variation of emissions. Diagnostic testing of municipal waste combustors (MWCs) has been carried out in order to discover how operating parameters affect emissions, especially those of polychlorinated dibenzo dioxins (PCDD) and polychlorinated furans (PCDF).

Recently the variable nature of ash residues has become a major issue in waste-to-energy facilities. The influences of various factors in the production, processing and management of ash residues can also be determined by methodologies similar those used in rationalizing waste combustion.

In order to be able to reduce undesirable emissions and properties of ash residues it is necessary to discover what factors influence them, how they can be measured and controlled, how they can be minimized, and what surrogates can be monitored to assure that optimum combustion and processing conditions are maintained on a continuous basis.

DIAGNOSTIC TESTING OF MUNICIPAL WASTE COMBUSTORS

The complex functions of a MSW combustion system can be diagnosed in order to ascertain the major factors and relationships which influence the emissions of concern.

One way to discern the effect of combustion parameters is to vary an individual parameter such as firing rate constant while running a series of tests in which another parameter, such as furnace temperature, is varied by adjusting combustion air. Another approach is to try a series of distinctly different settings of the dampers in the combustion system while holding firing rate and furnace temperature constant in order to discover which settings are more favorable. This procedure requires some way to decide what is most favorable or optimum.

Furnace volume and configuration, the amount of excess air, the placement of underfire and overfire air, and the degree of mixing of reactants, directly influence emission quantities and variations. The degree or tightness of control of fuel and air is another major factor controlling emissions and their variation.

Diagnostic tests of combustion systems thus include a matrix of conditions, including low, normal and maximum firing rates at a series of conditions applying various settings of excess combustion air, with subset runs with different balances of overfire and underfire air. These procedures have been applied at the Hamilton SWARU RDF-burning facility (2), the Quebec City mass-burn facility (3), and the Hartford RDF facility (4). The Prince Edward Island two-chamber starved-air facility was tested under four conditions, with low, normal and high secondary furnace temperatures, and with a longer charging cycle.(5) Tests of the Amager rotary kiln in Denmark were performed during startup and at various loads, as well as with sludge burning.(6)

The tests of the refractory furnace excess-air two-chamber MSW combustor at Pittsfield, MA were designed not only to cover the full range of furnace temperatures which could be sustained, but also to find the effect of various types of waste materials. The furnaces were supplied with municipal waste, commercial waste (PVC-free), and additional tests were run with added PVC and moisture.(7)

Correlation Matrices

When data from a large matrix of conditions is available, it can be analysed to determine the influence of various factors. Multivariate analysis reveals which parameters are most strongly related. Regression analysis can be used to obtain equations which describe the relationships between variables. The closeness of fit of the data to these correlations can be measured by regression coefficients.

The correlation matrix obtained from the Hamilton data showed that for runs under the normal damper settings carbon monoxide (CO), furnace temperature, underfire and overfire air, moisture in the fuel and in the stack gases, particulate matter, and chlorophenol in the stack gases correlated with a coefficient of 1.00 with polychlorinated dioxins (total PCDD).(8) Other factors such as oxygen, feed chute air, steam flow and excess air had coefficients from 0.97 to 0.92. However, the coefficients found when considering all 13 runs with various abnormal overfire air damper settings were substantially lower.

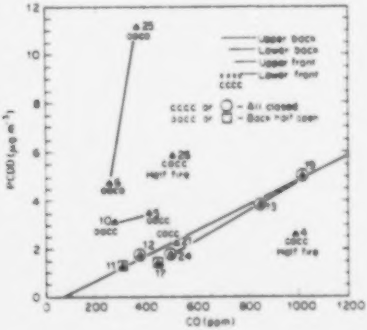

Figure 1. Emissions of Polychlorinated Dibenzo-dioxins (PCDD) versus Carbon Monoxide, from Tests of RDF Combustor at Hamilton, Ontario.(2)

The reason for these poorer correlations when data including the many trial damper adjustments is used becomes apparent when Figure 1 is examined. Tests with all overfire dampers closed except the lower back (COCC) show a linear correlation with PCDD. However, tests 6 and 25, with the upper back and lower front overfire air dampers open (OOCO), produced drastically higher PCDD. Test runs 5 and 10, with the upper back air open showed lower emissions. These data indicate that air from these sources interfered with combustion and produced higher dioxin emissions in spite of relatively low CO emissions.(8)

The Hamilton tests were performed under what is recognized today as very poor operating conditions. However, the obvious trends which promised reduced PCDD as CO was reduced indicated that CO would be a useful surrogate for improving combustion and reducing toxic organic emissions.

The environmental characterization tests of the Quebec water-wall municipal waste combustor (MWC) included five conditions, low, design, and high burning rates under good conditions, high and low radiant chamber temperatures and high and low excess air settings. Subset runs were carried out with different ratios of overfire to underfire air, and different ratios of back and front overfire air jets.

The correlation coefficients for operating variables showed that primary air gave a good correlation of 0.83, but total air only a fair correlation of 0.51.(3) On the other hand, dioxin correlations with monitoring variables showed the following correlations:

TABLE 1. CORRELATION COEFFICIENTS DETERMINED FROM QUEBEC TESTS

Parameter	Determination Coefficient	
	PCDD	PCDF
Copper	0.77 (very good)	0.72 (very good)
Cadmium	0.74 (very good)	0.75 (very good)
Zinc	0.70 (very good)	0.63 (very good)
Particulate	0.64 (good)	0.63 (very good)
Carbon monoxide	0.50 (fair)	0.60 (good)
Nitrogen oxides	0.38 (poor)	0.28 (poor)

The regression models also showed a close relationship between carbon monoxide and dioxins, furans, chlorophenols, chlorobenzenes, polyaromatic hydrocarbons (PAH) and to a lesser extent PCB's. This information again indicates that CO might be a good surrogate as an indicator of good combustion, and that keeping low CO serves to minimize emissions of all of these organic compounds. It also leads one to suspect that more easily measured organics such as CBs might also be practical indicators of trace organics such as dioxins and furans, which are costly and difficult to measure.

These correlation data are especially interesting because they indicate that particulate matter and the metals it contains are a major factor in the emissions of dioxins. While statistical correlations do not show direct cause and effect, nor explain the relationships, the strong relationship with particulate and metals is consistent with the findings of Vogg, Stieglitz and Hagenmaier, that dioxins can form on particulate matter when it is retained for long periods of time at the temperatures occurring in boilers, economizers and precipitators of municipal waste combustors. Formation was found to occur at temperatures ranging from 230° to 450°C, peaking at 300°C. While the formation occurs after the boiler, it requires carbonaceous material which is a product of poor combustion. For this reason it is not surprising that carbon monoxide, an indicator of good combustion in the furnace, correlates with dioxin formation in post-combustion regions.(9)

While statistical analysis reveals relationships, other methods must be used to discover and control the actual, extremely complex relationships between system design and operating parameters.

THEORETICAL RELATIONSHIPS BETWEEN VARIABLES

Calculation of Flame Temperature

Combustion variables such as theoretical flame temperature can be calculated from mass and energy balances, given the fuel composition and quantity, and combustion air quantity. Average furnace temperatures are determined by the amount of excess air or oxygen provided, and the moisture in the MSW, as shown in Figure 2. These temperatures, along with excess oxygen, control the formation and oxidation of carbon monoxide, some of which escapes unburned, and may also influence the amount of nitrogen oxides produced.

The factors determining flame temperature are thus the excess air or stoichiometric ratio (S.R.) and the moisture content of the waste. At less than stoichiometric air (starved air), the temperature produced is less than the maximum, and combustible gases are produced. As air is increased above a S.R. of 1.0, the excess air reduces the final temperature.

Combustion of MSW on a grate is much more complicated. Since it necessarily occurs in stages, including drying, volatization, combustion of volatiles, and combustion of carbon, many regimes of temperature occur. In the simplest system, the primary (underfire) air quantity supplied and the heat released by the fuel determines the average flame temperature prior to introduction of secondary air, and a second major regime of temperature occurs after secondary air is introduced. If the first stage is 'starved,' of air, secondary air may complete the combustion. If the combustion is essentially complete before secondary air is added, this air will reduce the gas temperature.

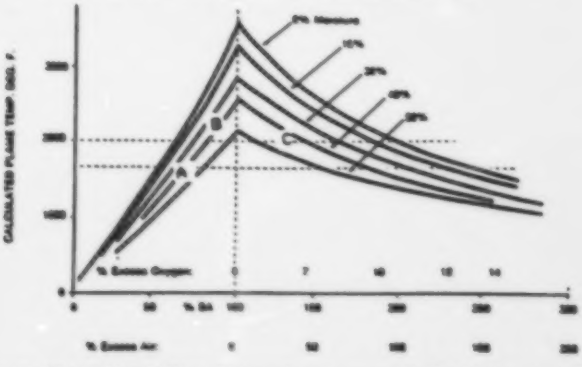

Figure 2. Theoretical Flame Temperatures versus Fuel Moisture and Excess Air and Oxygen.(10)

On a stoker grate, different parts of the fuel bed undergo a variety of conditions of fuel and air. When air is supplied in stages, even more complex conditions occur. There are an infinite number of paths to the final gas temperature. Points A and B in Figure 1 resulting in a final condition C illustrate this. Point A shows a primary zone temperature less than the final temperature, illustrating a typical starved-air two-chamber combustor. Point B shows the condition where the flame temperature exceeds the final temperature, which is typical of excess-air furnaces where secondary air is used to reduce flame temperatures. The connection between points A and B, and C may be anything within the envelope, and depends upon the manner of mixing of the gases.

Depending upon the heating value of MSW, which generally ranges from 4,500 to 5,200 Btu/lb, average flame temperatures may range from 2,200 to 2,500°F. In a waterwall furnace the volume required to cool these gases to 1,500°F before leaving the primary furnace results in a retention time of about 4 seconds.(10)

Thermal Destruction of Organics

The destruction of organic gases or vapors is essentially instantaneous in a flame, and non-flame temperatures exceeding about 1500'F have been shown to destroy the toxic organic compounds of concern, provided that a few percent of excess oxygen are present in all locations and a retention time of about one second is provided.(11) To be certain that no pockets of gas get by without meeting these conditions, either better mixing or some excess temperature and oxygen are generally provided.

On the other hand, solid particulate which may be carried by the combustion gases behaves differently, carrying partially burned organics out of the flame and the combustion chamber. These are probably the primary vehicle for escape of organics, as well as later formation of dioxins and furans.

RELATIONSHIPS DETERMINED BY ACTUAL TESTS

Test data obtained from tests of full-scale MWCs show consistent relationships between certain parameters, such as oxygen and furnace temperature, and carbon monoxide and temperature or oxygen, which can be continuously monitored and logged. When plotted against each other, the data points from a series of operating conditions fall close to a correlation line.

Oxygen and temperature are found to have a linear relationship, whereas carbon monoxide exhibits a logarithmic relationship with temperature or oxygen, reflecting a logarithmic approach to zero CO as temperature rises, or oxygen falls.(12) Likewise, NOx generally increases (slowly) with increased temperature. However, NOx does not decrease with temperature below a level which apparently reflects the nitrogen content of the MSW. NOx shows a seasonal variation due to changes in MSW nitrogen content, however.

A serious difficulty in finding relationships between combustion parameters and emissions, especially of trace organics such as PCDD and PCDF, is the length of time required to collect samples of sufficient quantity to obtain measurements of such trace components. Tests of four to 12 hours may be needed to obtain sufficient quantities to exceed the detection limits of analytical devices. Hence, PCDD and PCDF data represent average conditions over these long periods, during which substantial variations and fluctuations have taken place. Variations in results between tests performed under supposedly identical conditions of the control variables reflect the stability of control which was maintained, as well as secondary variables.

Carbon Monoxide as a Surrogate for Good Combustion

Carbon monoxide (CO) measured in the products of combustion has been found to be a useful indicator of success in destroying organics since CO is created by the flame and is more difficult to convert to CO_2 than the other organic compounds, including methane.

The extent to which CO measurements relate to the emissions of critical toxic compounds such as PCDD and PCDF has been a matter of much debate, since the phenomena involved are complex, and related both to the combustion process and the post-combustion regions of the boiler. However, data from tests of numerous facilities show the same general linearity between CO and total PCDD+PCDF. Tests of the rotary kiln MWC in Amager, Denmark also demonstrate linearity between the Toxic Equivalent 2,3,7,8 TCDD and carbon monoxide while the furnace was operated under startup, shutdown low and high firing rates and even sludge burning.(8)(12)

Furnace Temperature and Oxygen as Control Parameters

The variations in PCDD and PCDF with combustion parameters such as temperature, oxygen and CO are remarkably similar, although the relationships are not sharply defined. Increasing temperature, reducing oxygen, and reducing CO all serve to reduce PCDD and PCDF up to a certain point.(12) Therefore controlling oxygen is an alternate method of controlling temperature, although the temperature will then vary with the moisture content of the MSW.

Figure 3 shows the PCDD plus PCDF data obtained at Pittsfield plotted versus secondary furnace temperature for 8 separate tests wherein the waste composition and operating conditions were varied. Each test condition was repeated, with all conditions maintained constant. In spite of this, a wide divergence was found between supposedly duplicate runs. If a large number of test runs had not been undertaken, it would not have been possible to establish trends in the data. Figure 3 shows carbon monoxide plotted versus temperature for comparison with Figure 4. It is apparent that CO and PCDD+PCDF have similar trends with furnace temperature, and that on logarithmic paper the data tend to follow straight lines. However, although the lines point toward zero as temperature rises, emissions increase above about 1700°F, measured after complete mixing of secondary air.(12)

Two phenomena require investigation in these graphs: the dispersion between pairs of duplicate test points in PCDD+PCDF data, and the increase noted with increased temperature.

Examination of data from Pittsfield in the region where CO is low (below 50 ppmv or so), shows wide scatter, as noted above. Similar increases in PCDD and PCDF were noted in PEI data at low CO. When the background data for these duplicate tests are examined in search of other variables, it is found that MSW moisture is a factor. It is noted that the higher PCDD/DF points represent tests with higher moisture and lower points have lower moisture content. Thus it appears that moisture influences either the PCDD+PCDF or the CO readings, or both.

At temperatures above the 1700°F point the Pittsfield data show that the lowest CO readings correspond to the highest PCDF emissions, indicating that another factor, probably shortage of oxygen, causes furans, rather than dioxins, to form.

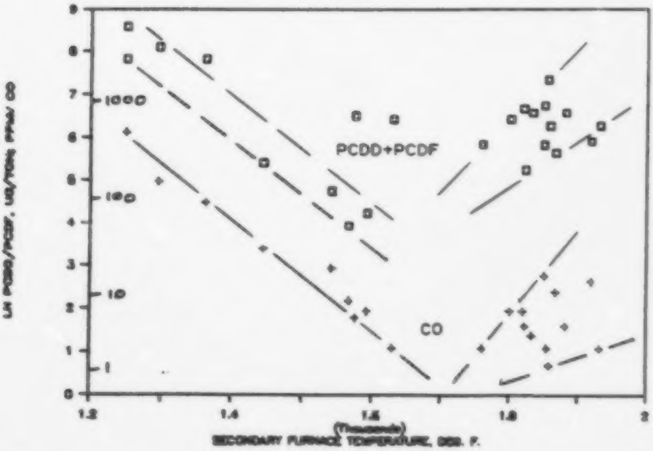

Figure 3. Carbon Monoxide (CO) and Total Dioxins and Furans versus Furnace Temperature After Addition of Secondary Air.

The Pittsfield tests have provided the most comprehensive set of data on PCDD and PCDF emissions measured at the boiler outlet. Figure 3 shows both carbon monoxide (CO) and total PCDD+PCDF emissions versus the secondary furnace temperature (fully mixed after secondary air has been added). The data are plotted as micrograms per ton rather than nanograms per cubic meter (ng/Nm^3), showed a somewhat more consistent correlation. The logarithm of CO emissions data falls on a straight line throughout the wide range of secondary furnace temperatures, pointing to a virtual minimum at 1700°F, indicating logarithmic decay. At higher temperatures, a leveling off, or a sharp rise is found. The highest rise is noted for tests with dry commercial waste, waste with PVC added, or tests with low oxygen. The lowest emissions were from tests with water added to the waste.(12)

These data indicate that efforts to achieve temperatures (after secondary air has been mixed in) of 1800 F or higher may be counterproductive from the standpoint of dioxin destruction. The context of these tests should be understood: the Pittsfield MWC uses flue gas recirculation as a means to adjust and maintain furnace temperatures, in addition to adjusting excess oxygen. As temperatures were increased above 1750°F, excess oxygen was reduced from 9% to as low as 6.5%, due to limitations on combustion air capacity. This caused more flue gas recirculation to be applied in order to hold down the temperature. The high moisture MSW did not call for as much recirculation, but suffered the same drop in excess oxygen with increased temperature.

Comparison of Pittsfield and Quebec Test Data

Figure 4 shows the relationship between the logarithm of total PCDD+PCDF and furnace temperature. The hollow points show the data obtained during the Pittsfield tests, and the solid points the data from the Quebec City Emission Characterization Tests. In both cases the temperatures were measured after secondary air injection. The data from both sources, one a refractory furnace, the other having a waterwall furnace, fall within the same range.

Poor operating conditions of the Quebec MWC produced total PCDD+PCDF in the 500 to 1000 ng/Nm^3, whereas good conditions produced emissions from 40 to 300 ng/Nm^3.

The range of emissions at any given temperature in both cases was generally about one order of magnitude (ten times), showing that temperature alone is not the only variable.

The Pittsfield data include points ranging from 900 down to 13 ng/Nm^3. The points with high moisture fuel lie at the lower boundary of the range, whereas the points representing runs with large amounts of PVC added are on the upper boundary. The data show that the high emissions at temperatures over 1800°F reflect greatly increased furans, hence may be the result of lack of oxygen at higher temperatures.

There is a remarkable correspondence between the Quebec City data and the Pittsfield data, in spite of the difference in technology. The Quebec data represent five operating conditions (15 data points), whereas the 24 separate test runs at Pittsfield represent 12 different conditions, and hence provide a more detailed continuum of data. The combination of the data enriches the interpretation of both tests.

The logarithmic relationship between both CO and dioxin emissions with temperature shows these are functions of chemical kinetics. The increase in emissions at higher temperatures are associated with sharp spikes in CO readings, indicating poor mixing of fuel with oxygen.

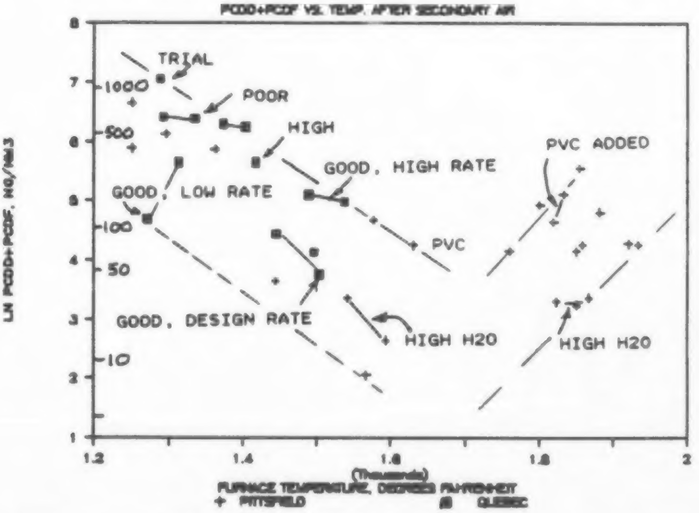

Figure 4. Total PCDD+PCDF versus Furnace Temperature After Injection and Mixing of Secondary Air, Pittsfield and Quebec Data.(3)(8)(12)

Underfire and Overfire Air Ratios

The Quebec tests specifically varied underfire and overfire air ratios, at both high and low excess air, to ascertain the effect of these factors on organic emissions. The results, seen in Figure 4, show that low excess air and higher ratios of underfire to overfire air produced the lowest PCDD/PCDF emissions.

Simply looking at these ratios does not reveal a factor which may be more important. Figure 5 shows the calculated stoichiometric ratio (S.R.) of the underfire air versus PCDD+PCDF. The data for Good Combustion fall on a line which points toward a S.R. of 0.8, and the optimum operating condition occurred at a S.R. of 0.9. Higher ratios resulted in substantially higher organic emissions. This graph gives evidence that best combustion occurs when undergrate air has approximately zero excess air. It must not be overlooked that this is an average over the entire grate, hence local conditions will include starved air and excess air regions. In any case, it is evident that starved air conditions are favorable in the region of rapid combustion, limiting surges in combustion, minimizing lifting of flaming material, preventing excessively high temperatures, and producing volatile gases to be burned above the grate.

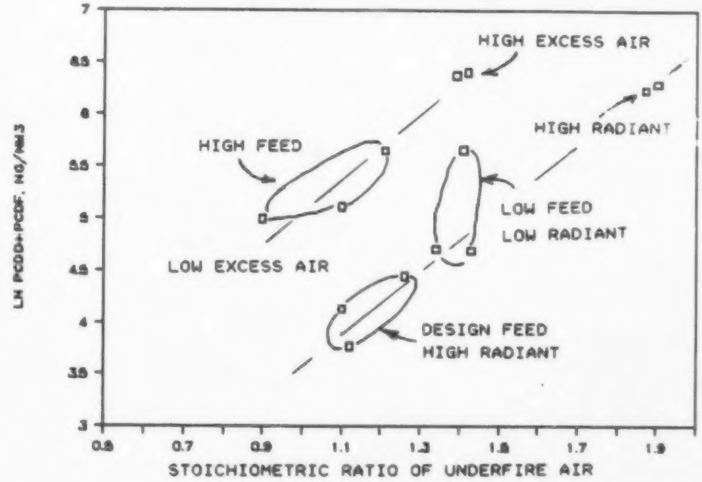

Figure 5. Logarithm of total PCDD+PCDF versus Stoichiometric Ratio of Underfire Air, calculated from Quebec data.

CONTROLLING COMBUSTION

In order to control combustion and maintain optimum conditions, it is necessary to measure some parameter and use this measurement to actuate controls such as feeders and combustion air dampers. Furnace temperatures have been used, traditionally, and oxygen controls have been found to give stable control. However, furnace temperatures are difficult to measure, since the desired temperatures are destructive of the sensing devices. To avoid this problem, sensors are located at the outlet of the furnace, or beyond, so that actual flame temperatures are not measured. New temperature-measuring devices are being tried, with the objective of obtaining direct and instantaneous flame temperature readings. However, the problem remains that optimum conditions are probably a combination of temperature, moisture and oxygen, together with variable distribution of overfire and underfire air. This complex may be solved by computer logic.

Multivariate analysis reveals the contributing factors, but not the reasons for their influence. For instance, dioxin emissions are found to be related to firing rate, particulate emissions, excess oxygen, lower furnace temperature, nitrogen oxides and moisture. Most of these variables are directly dependent on each other, and these dependencies can be related by relatively simple calculations, given the chemical composition (including moisture content) and feed rate of the waste, the air flow, and the excess oxygen. If flue gas moisture were measured as well as oxygen, the control computer could calculate the theoretical flame temperature.

Furnace design and configuration

The furnace design and configuration, including surface, volume and shape, as well as location of overfire air injection points, greatly affects combustion performance. The rate of temperature drop as combustion products pass through water-cooled furnaces and boilers can be calculated knowing the thermal absorption characteristics of the furnace and waterwalls, along with the furnace volume which determine the gas temperatures leaving the furnace. The amount of furnace surface, hence volume, which is needed to drop gas temperatures low enough to prevent damage of boiler tubes (1200 to 1500°F) inherently provides retention time for the organics to continue to become oxidized. Good mixing reduces the time for the completion of reaction in all locations. No amount of time will eliminate streaks of gas which are too rich or too lean, unless mixing action is applied by the configuration. Refractory furnaces sustain the temperatures, hence may not need as high an initial temperature to achieve the same effect as a water-cooled furnace.

The amount of particulate lifted from the burning grate into and out of the furnace is strongly dependent on the gas velocity, hence on the firing rate of a given furnace configuration. Excessive lifting velocities and inadequate furnace volumes result in particles leaving the furnace before completely burning out. Such carbonaceous particles can become sites for dioxin formation and emissions of other organics.

CONTINUOUS EMISSION MEASUREMENT

Continuous emission monitors, especially oxygen and carbon monoxide readings, provide important indications of the quality of combustion and of emissions. Due to the variations in these readings, statistical methods are useful tools for appraisal of the data, and to determine what adjustments are effective in optimizing combustion and minimizing emissions.

Combustion fluctuations

Continuous recordings of oxygen and CO readings gives important information about the relationship between these variables. When simultaneous readings are plotted against each other, a general relationship is found, as shown in Figure 6. These data, from Pittsfield tests, show a wide range of operation and specific areas representing tests where temperature was held constant within about 20°F. The variation in CO is the result primarily of excess oxygen, but the effectiveness of mixing and also of the presence of moisture can be expected to change the CO emissions even at constant temperature.

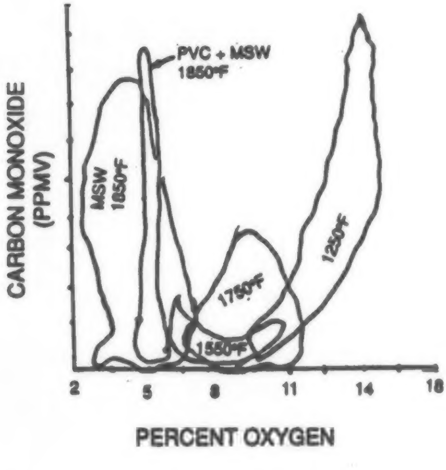

Figure 6 Envelopes of CO and Oxygen Continuous Emission Data from Pittsfield Tests at Various Constant Furnace Temperatures.

Another factor, harder to discern, is the degree of mixing of fuel and oxygen. The importance of mixing increases as available excess oxygen decreases. It is evident from Figure 6 that below about 6% excess oxygen unstable conditions have taken place, with spikes in CO. Continuous graphs of CO during these tests show the presence of CO spikes under both the low and high temperature conditions. These can be attributed to pockets of poorly-mixed reactants, where the air/fuel ratio departs grossly from the optimum. The lower the amount of excess oxygen, on average, the greater the probability of local deficiencies, and the greater the need for high-energy mixing effort.

Control systems which attempt to maintain either furnace temperature or oxygen improve the consistency of operation, but are still subject to oscillations which create some variability in emissions. When continuously recorded data is analyzed statistically, the variation is found to have the familiar bell-shaped curve reflecting random conditions. The range of variation, which can be defined by calculating the standard deviation, conveniently reflects the 'tightness' of control. Of course the range of control can be judged directly by looking at recorder trend charts, such as Figure 7. This graph shows an upset, and reveals the amount of time the oxygen control system took to return to normal operation after an interruption in feed (about 15 minutes).

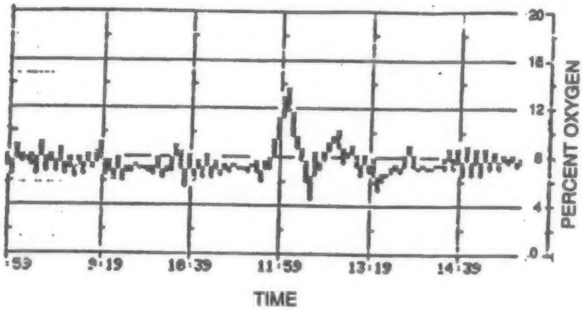

Figure 7. Variations in Oxygen Measured at the Furnace Outlet by a Continuous Monitor Trend Recorder.

Distribution of oxygen and carbon dioxide data

The control system of a MWC has the function of adjusting combustion air to suit the amount of fuel being oxidized at a given moment. Whether the control sensor measures oxygen or whether temperature is used to control combustion air, there will be fluctuations in the actual excess oxygen, and consequently in the amount of CO produced.

The histogram shown in Figure 8 represents the variations in excess oxygen readings averaged for one hour, over the course of 100 hours' operation of a MWC. The peak is between 10.5 and 11% oxygen, dry basis, which corresponds to about 8.5 to 8% oxygen on a wet basis. The distribution is symmetrical about the mean, except for a separate population of oxygen readings close to 16% representing readings taken during a weekend shutdown when auxiliary fuel was used to keep the furnace hot.

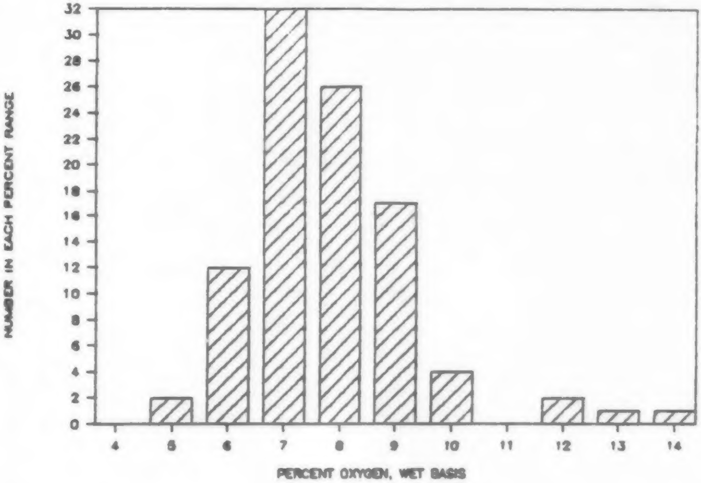

Figure 8. Histogram of Oxygen Readings

The corresponding CO data in Figure 9 show a skewed histogram, with characteristic of log-normal distributions. The CO readings show less sensitivity to excess oxygen. About 3% of the data show about 93 ppm of CO, which was characteristic of firing with auxiliary fuel when MSW was not being burned.

The standard deviation and coefficients of variation of data obtained from continuous emission monitoring can be used as an indication of the quality of the combustion control system. In this case, the s.d. of the CO data, averaging 29 ppmv, was 17.5 ppm, resulting in a coefficient of variation (CV) of 60%. The corresponding oxygen data averaged 12.1%, with a s.d of 2.6 and a CV of 22%. Evidently the CO varied much more than the oxygen.

To the extent that CO varies linearly with PCDD/PCDF, it can be expected that the variation in dioxins and furans will be roughly the same as the CO variation, increasing logarithmically as temperature falls or excess oxygen increases.

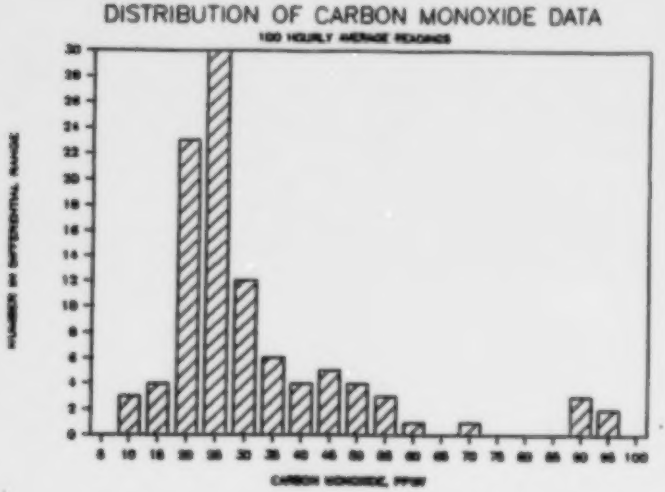

Figure 9. Histogram of Carbon Monoxide Readings Averaged Over 100 Hourly Intervals, Showing Symmetry of the Logarithm of CO.

ESTIMATION OF PROBABLE EMISSIONS FROM PLANNED FACILITIES

Estimating average and maximum upset emissions

An estimation of maximum emission rates, and their relationship to the average or mean, can be made by calculating the mean plus two standard deviations, which includes 95% of the data. In this case the CO reading which is not exceeded by 95% of the data is 2.2 times the mean. None of the 100 data points exceeded 93 ppm, or 3.2 times the mean. In other words, the maximum upset CO reading in 100 hours was 3.2 times the average, and lasted for less than 1% of the time period. A maximum value of 64 ppm was exceeded for only 5 hours out of 100.

Log Probability Graphs

It has been found that the emissions reported from a series of tests of a given facility, or from tests of a number of similar facilities generally follow a log-probability distribution, that is, data plotted on these coordinates fall close to a straight line.(1)(10) Figure 10 shows data from 14 MWCs having lime scrubbers and baghouses as emission control devices. This graph was plotted by sorting the data in ascending order, then plotting the points equally spaced on linear coordinates. This is a simpler plot than using probability paper, but the results are almost the same, except for upper and lower tails.

The range of variation in Figure 10 is largely due to the different scrubbing temperatures, since these temperatures control the amount of trace organics which are condensed on collected particulate. These data show that the 2,3,7,8 TCDD Toxic Equivalents (EPA) did not exceed 0.3 ng/nM^3, and averaged about 0.12 ng/nM^3. The maximum value was less than 3 times the mean. The Swedish goal for T.E.(Eadon) is 0.1 ng/nM^3; this is equivalent to a T.E.(EPA) of about 0.05 ng/nM^3, which was met or exceeded by three of the plants.

It should not be overlooked that post-combustion emission controls employing temperature reduction to the vicinity of 300°F with suitable particulate controls will capture essentially all organic emissions, regardless of the input quantities. The use of these controls reduces the requirement for achieving minimum emissions from the combustion process. However, the organics will then reside in the collected particulate matter, increasing concern that it be disposed of properly.

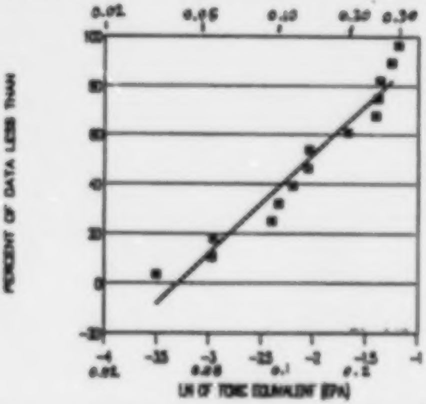

Figure 10. Toxic Equivalent Dioxin Emissions from Municipal Waste Combustors Equipped With Acid Gas Scrubbers.

THE IMPORTANCE OF MIXING IN ASH RESIDUE MANAGEMENT

The management of flyash and bottom ash residues from combustion of MSW requires attention to basic principles in the same manner needed in management of its combustion. The composition and leaching properties of flyash, scrubber residues and bottom ash residues from combustion of MSW are highly variable, and dependent not only on the composition of the MSW, but also on the conditions to which the waste and its products have been subjected. Statistical analysis shows that these residues normally exhibit consistent and random properties, but that higher variability can result from poor wetting and mixing of these different types of residues. The variations in ash properties obtained from a series of tests may be a direct and useful indication of the mixing effectiveness of materials handling operations.(1)

Failure to mix residues properly has led to highly variable test data, resulting in public concern about the toxicity of ash residues. Flyash generally contains higher concentrations of soluble heavy metals than bottom ash, since metals such as lead and cadmium condense on flyash surfaces in the soluble oxide form or become soluble chlorides. The solubility of certain forms of these metals is highly dependent on the pH of leaching water. If flyash is not well mixed with bottom ash, samples may contain abnormally large fractions of flyash and consequently exhibit high concentrations of lead and cadmium when subjected to leaching tests.(1)(12)

When lime scrubbers are used, the lime will tend to cause the lead and cadmium to be converted into insoluble compounds or complex cementicious formations in the presence of sufficient moisture. However, if mixing and moisture addition are not uniform, a larger fraction of these metals may remain in water-soluble forms, and may thus fail leaching tests. The unique physical and chemical properties of flyash, scrubber products and bottom ash offer potentials for reuse to avoid landfilling, and in any case require careful handling so that they do not harm the environment.

CONCLUSIONS

As a result of extensive research on the relationship between refuse composition and combustion conditions on the quantity and nature of emissions, it has been found that, while refuse composition varies widely, emissions from combustion are consistent in their variability, and range within rather narrow limits when modern combustion control devices are applied.

In addition to providing sufficient furnace volume to complete combustion, it is necessary to find and maintain the range of excess oxygen (or corresponding furnace temperatures) which achieve optimum combustion, the optimum balance between underfire and overfire air, the best injection points, and the optimum configuration of high-intensity mixing of the fuel and air.

In order to encourage environmentally sound disposal and reuse of ash residues, attention to proper management is required, much as fuel and combustion air must be properly controlled to minimize emissions.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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COMBUSTION CHARACTERIZATION OF RDF INCINERATOR TECHNOLOGY: A JOINT ENVIRONMENT CANADA - UNITED STATES ENVIRONMENTAL PROTECTION AGENCY PROJECT

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ABSTRACT

Environment Canada and the U. S. Environmental Protection Agency are conducting a joint project to evaluate pollution and pollution control technologies associated with refuse derived fuel (RDF) facilities. This project involves comprehensive field tests at the Mid-Connecticut (Mid-Conn) waste-to-energy facility owned by the Connecticut Resource Recovery Authority. The facility incorporates a municipal solid waste processing plant and three spreader-stoker RDF combustors. Emissions from each combustor are controlled by a lime spray dryer and fabric filter baghouse.

A primary objective of the Mid-Conn test program is to develop correlations between combustion process conditions and furnace emissions. Adjunct activities include evaluation of the flue gas cleaning (FGC) equipment and combustion facility residues. The field test project is divided into two test phases: characterization tests and field tests. This paper discusses the result of the combustion evaluations conducted during the characterization test phase.

This paper has been reviewed in accordance with the U. S. Environmental Protection Agency peer and administrative review policies and approved for presentation and publication.

INTRODUCTION

In June 1987 Environment Canada (EC) and the U.S. Environmental Protection Agency (EPA) implemented interagency agreements to conduct joint programs related to municipal waste combustion (MWC). The objective of these agreements was to combine technical and financial resources to develop information to regulate MWC facilities and identify appropriate pollution control technologies. These agreements cover three major program elements: (1) combustion control; (2) flue gas cleaning (FGC) technologies; and (3) residue characterization and treatment.

The single largest project under the joint MWC program is the field test evaluation of the Mid-Conn Resource Facility in Hartford, Connecticut. This facility features a municipal solid waste (MSW) processing plant and a boiler plant with three refuse derived fuel (RDF) combustors, each rated at 231,000 lb/hr* of steam. Furnace emissions from each unit are controlled by a lime spray dry adsorber (SD) and fabric filter baghouse (FF).

The Mid-Conn test project is divided into two test campaigns, characterization tests and performance tests. During the characterization tests, combustion and FGC process conditions are set and process conditions, furnace emissions, and stack emissions are measured. Measurements on these tests are generally limited to those obtained from process instruments and continuous emission monitors. The different conditions evaluated during the characterization tests provide information needed to define a limited number of test conditions for the performance tests. The performance tests include intensive sampling and analyses needed to measure process conditions and to characterize all input and output streams from the combustion facility. Emissions measurements for the characterization tests included those for oxygen (O_2), carbon dioxide (CO_2), carbon monoxide (CO), total hydrocarbons (THC), sulfur dioxide (SO_2), hydrogen chloride (HCl), and nitrogen oxides (NO_x). The parametric test includes measurement of these parameters and sampling and analysis for trace organics, particulate matter, and trace metals.

The primary objective of the Mid-Conn project is to develop correlations between combustion process conditions and furnace emissions in RDF combustors. Adjunct activities on the project include the development of correlations between FGC process conditions and emissions, and the characterization of residues from the RDF combustor and FGC system. This paper discusses the combustion portion of characterization tests and identifies test conditions selected for the performance tests.

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*English Engineering Units rather than metric units are used here. See Appendix for unit conversions.

BACKGROUND

In 1984 EC initiated its National Incinerator Testing and Evaluation Program (NITEP) with a comprehensive field test of a modular starved air incinerator on Prince Edward Island.[1] In 1985 NITEP pilot plant tests to evaluate the use of dry scrubbing and wet-dry scrubbing to control emissions from mass burn incinerators were completed.[2] In 1986 NITEP tests were conducted at the Quebec City Urban Community Incinerator to evaluate the use of improved combustion conditions for control of emissions from conventional waterwall mass burn incinerators.[3] Based on these NITEP test projects and other activities, the Canadian Council of Resource and Environment Ministers (CCREM) issued Operating and Emission Guidelines for Municipal Solid Waste Incinerators in October 1988.[4]

In 1986 EPA began a comprehensive study of MWC. This study culminated in a nine-volume report with a summary "Report to Congress."[5] This study, along with other activities, resulted in an advanced notice of proposed rulemaking (ANPRM) by EPA, announcing the development of new rules applicable to existing and new MWC facilities in the U.S.[6] These rules are to be proposed in November 1989 and promulgated in December 1990. In developing these new rules EPA is considering the application of good combustion practice (GCP) and appropriate FGC technology. One of the FGC technologies being evaluated by EPA is spray dry adsorption used in conjunction with fabric filtration. Recommendations for GCP in mass burn, modular starved air, and RDF combustors were documented in the EPA report "Municipal Waste Combustion Study: Combustion Control of Organic Emissions."[7] The GCP recommendations were based in large part on discussions with MWC designers/operators and on the results of the NITEP combustion tests at Prince Edward Island and Quebec City.

In 1987 EC and EPA initiated a joint program to evaluate methods of controlling pollution from MWC facilities. Two key objectives of this joint program were to (1) conduct comprehensive combustion characterization tests at an RDF facility and (2) conduct full scale parametric performance tests on a spray dryer fabric filter system at a MWC facility. The tests at the Mid-Conn facility meet both of these objectives.

The Mid-Conn project is jointly funded and directed by EC and EPA. Contractor support is provided by firms in Canada and the U.S. Energy and Environmental Research Corporation of Irvine, CA, was responsible for selection of the Mid-Conn test site and developing the initial project test plan. Alliance Technologies Corporation of Bedford, MA, is responsible for execution of the characterization and performance tests.

TEST FACILITY DESCRIPTION

The Mid-Conn facility is owned by the Connecticut Resource Recovery Authority. The facility will receive MSW from more than 23 communities in the Hartford, CT, vicinity. The facility was designed and constructed by Combustion Engineering. The facility contains a MSW processing plant, provisions for coal receiving and storage, RDF transport conveyors, and a boiler house with associated air pollution and residue handling equipment. The facility is capable of burning either RDF or coal.

The RDF processing plant is designed to operate at a full load of 100 tons/hr on each of the two processing lines. MSW is dumped from trucks onto a tipping floor. The MSW is inspected and bulky items and potentially hazardous materials are removed. The waste is then moved by front-end loaders to a flail mill. After passing through the flail mill, iron and steel are removed by magnetic separators. The remaining waste is then conveyed to large rotary trommel screens, which allow non-combustible residue such as glass and sand to be removed. The second stage of the trommel separates out the combustible fraction which does not need further size reduction. Oversize material is conveyed to a hammermill shredder for final size reduction. RDF is stored in piles in an enclosed storage area and is transported via conveyors to the boiler house as needed.

The boiler house contains three CE steam generating units. Each unit includes a natural circulation welded-wall boiler with 2-1/2 in. OD tubing on 3-in. centers (see Figure 1). The superheater consists of a vertical two-stage pendant platen design which is located over the furnace's nose arch to protect it from direct radiation. Following the superheater is an economizer which consists of two horizontal banks of in-line tubes. A tubular combustion air preheater is also incorporated into each unit after a long duct run on the far side of the turbine room adjacent to the spray dryer.

The fuel burning system consists of a CE Refuse Combustor Stoker and specially designed grate (see Figure 2). The grate includes a self-cleaning key design to remove fused or clinkered ash during grate operations. Multiple undergrate air zones provide controlled air flow to left- and right-hand areas at five separate zones along the grate length. The fuel is conveyed from the RDF storage area to surge bins located at the front of each boiler. A vibrating pan feeder is used to feed the fuel uniformly to the stoker.

The overfire air system is separate for the coal and the RDF burning. The RDF system consists of four tangential overfire air (TOFA) windbox assemblies located in the furnace corners. Each of the TOFA windbox assemblies contains three elevations of nozzles which can be manually adjusted in the horizontal plane. Preheated air is admitted tangentially to form a vortex. Overfire air ports for coal combustion are located on the front and rear walls. One row of ports is provided on the front wall and two rows on the rear wall. Thus overfire air can be provided in a tangential mode, a wall-mode, or a combination of both modes.

The ash system removes ash in two streams which are later combined and stored for eventual disposal. The first stream contains the bottom ash, economizer ash, and stoker siftings. The second stream collects baghouse and air heater ash. The latter ash stream is conditioned in a pug mill before it is combined with the first stream. The combined streams are transported to a three-sided storage bin.

The flue gas cleaning system consists of a lime-based spray dryer followed by a reverse-air fabric filter. The spray dryer is capable of controlling the temperature at the baghouse inlet and SO_2 concentration at the baghouse outlet. The baghouse inlet temperature is controlled by the amount of lime slurry and dilution water flow rates. After the temperature is set the SO_2 removal rate is controlled by adjusting the lime slurry flow rate. Each baghouse has 12 compartments, each with 168 Teflon-coated fibrous glass bags.

COMBUSTION TEST OBJECTIVES

A major objective of the characterization and performance tests is to identify optimum combustion conditions and establish the boundaries between good and poor combustion. Another major objective is to test hypotheses concerning formation and destruction of chlorinated dibenzo-p-dioxins and chlorinated dibenzofurans (CDD/CDF).

A basic premise of the EC/EPA incineration programs is that high emission of CDD/CDF from MWC facilities is related to poor combustion and that good combustion results in low CDD/CDF emission. Good combustion can be achieved by appropriate design and operating practices. The elements of good combustion include appropriate combustion temperatures, the proper amount and distribution of combustion air, mixing of waste thermal decomposition products with combustion air, and sufficient time for the completion of mixing and combustion reactions. The recommended criteria published by EPA in 1987 and CCREM in 1988 for defining good combustion are similar[7,4].

The characterization test matrix was devised to investigate the effects of load, excess air, and air distribution on combustion conditions as defined by CO emissions, and by particulate matter carried out of the combustor (PM carryover). Low CO concentrations in flue gases provide one of the best indicators of good combustion. The oxidization of CO to CO_2 is the final step in the hydrocarbon chain and, if high levels of CO are present in the exhaust gases, it is probable that significant quantities of organic compounds are also present. High emissions of CO in MWC systems maybe due to a number of conditions:

- insufficient overall amounts of combustion air
- excessive overall amounts of combustion air leading to low combustion temperatures
- poor mixing resulting in localized fuel rich regions or localized

- excessive carryover of particulate bound organics from the furnace into the colder regions of the combustion system before burning is complete.

An evaluation of tests at Quebec City and elsewhere indicates that CDD/CDF emissions are highly correlated with PM carryover.[3,8] It is postulated that the amount and properties of particulate matter carried out of the furnace largely determine the amounts of CDD/CDF which are believed to be created by catalytic reaction with flyash in the temperature range of 482 to 662°F (250 to 350°C). Downstream formation can be controlled by destroying organic material before it leaves the furnace and by minimizing the amounts of particulate matter, both organic and inorganic, leaving the furnace as flyash. Spreader stoker RDF combustors have inherently higher amounts of PM carryover than modular or mass burn MWC systems. A primary objective of the combustor tests is to evaluate methods of minimizing PM carryover and determine the relationship between carryover and downstream formation of CDD/CDF in the air preheater at Mid-Conn. The temperature at the economizer outlet on Unit 11 typically ranges from 650 to 700°F while the flue gas temperature at the outlet of the air preheater ranges from 350 to 400°F.

TEST INSTRUMENTATION AND METHODS

Continuous emission monitors (CEMs) were used to measure gaseous CO, O_2, CO_2, HCl, SO_2, NO_x, and total hydrocarbons. These monitors were calibrated using EPA- and EC-approved methods. Manual methods were also used to determine the moisture, CO_2, O_2, and HCl content in the flue gas. Moisture content in the gas entering the spray dryer absorber was also determined with a non-dispersive infrared monitor. Gas flow rates at the sampling train locations were determined using pitot tube traverses. The steam flow, air flow, pressure, and temperature data for the combustor were obtained from plant instrumentation. Other pressure and temperature data were measured using equipment installed for the tests. A data acquisition system acquired real time data and processed it into the desired form. The sampling trains were checked and used in accordance with EPA- and EC-approved protocols.

Data for each test were collected after the combustion conditions were established. The plant operator adjusted the combustor to conform with the desired test conditions and the steam load, excess O_2, CO, and other instrumentation was monitored. When these combustion and FGC conditions were stabilized the test was begun. The CO value measured at the spray dryer absorber was used as the main criterion for good or poor combustion.

CHARACTERIZATION TEST CONDITIONS

The planned combustion characterization tests were divided into five categories or series which were run to determine the effects of the following operating conditions on emissions:

Series A: Undergrate to overfire air (UGA/OFA) combinations at full load and normal excess air.

Series B: UGA/OFA combinations at part load and normal excess air.

Series C: UGA/OFA combinations at peak load and normal excess air.

Series D: Excess air levels.

Series E: Undergrate air distribution.

A summary of the characterization test operating conditions and the resulting O_2 and CO emissions are presented in Figure 3 for Test Series A, Figure 4 for Test Series B, and Figure 5 for Test Series C and E. Test Series D was deleted after several unsuccessful test attempts because of the difficulty of running the combustor at a steady steam load while holding excess air constant and maintaining stable CO emissions.

The numerical values given in Figures 3 through 5 represent test averages. The O_2 and CO concentrations were measured at the inlet to the spray dryer adsorber. The O_2 measurement is correct to 12 percent CO_2. The furnace temperature is calculated from steam flow conditions and furnace flue gas measurements.

SUMMARY OF RESULTS

The characterization tests indicate that CO emissions are very sensitive to excess air levels (O_2), combustion air distribution, and load. During the full load tests (Series A), average loads ranged from 216,000 to 232,000 lb/hr steam flow. The average O_2 concentrations measured at the spray dryer inlet ranged from 7 to 11 percent, and the average CO concentration at the same location ranged from 52 to 455 ppm (see Figure 3). The best combustion conditions at full load as defined by CO were achieved using three levels of TOFA (3-TOFA) in combination with the maximum amount of rear wall air (35-RWA). Average CO emissions for this characterization test (CT-12A) were 52 ppm. The average steam load and O_2 concentration during CT-12A were 231,000 lb/hr and 8 percent. The next best test at full load (CT-08) with average CO emissions at 73 ppm, was achieved with two levels of TOFA (2-TOFA) and 35-RWA. The highest average CO emission (455 ppm) was recorded during CT-13 with one level of TOFA (1-TOFA) and 35-RWA. During this test the load averaged 232,000 lb/hr of steam and the flue gas oxygen content averaged 7 percent. It is likely that the high CO levels measured during test CT-13 resulted from low O_2 levels as well as poor mixing.

Loads during the low load tests (Series B) ranged from 143,000 to 192,000 lb/hr while O_2 levels at the spray dryer inlet ranged from 11 to 17 percent. (see Figure 4). Average CO emissions tended to be substantially higher during the part load tests than from the full load tests. Average CO emissions measured at the spray dryer inlet varied from 151 to 952 ppm. The test with lowest average CO emission (CT-5) was conducted with 1-TOFA and at an average load of 153,000 lb/hr and an O_2 concentration of 12 percent. The high CO_2 emissions during low load operation probably resulted from a combination of poor mixing and reduced furnace combustion temperatures.

Four high load tests (Series C) were run at loads ranging from 240,000 to 246,000 lb/hr steam flow (see Figure 5). Oxygen concentrations ranged from 6 to 7 percent oxygen, and CO concentrations ranged from 381 to 426 ppm. It is believed that the high CO emission levels during the high load tests resulted more from low overall O_2 levels than from poor mixing.

The UGA distribution effects tests (Series E) were run at steam loads ranging from 207,000 to 231,000 lb/hr. Average O_2 concentrations ranged from 11 to 7 percent while average CO concentrations ranged from 106 to 385 ppm (see Figure 5). Unit 11 is normally operated with the undergrate air dampers in the positions designated as II (15, 90, 90, 90, 30). While there is no indication that any other UGA distribution provides lower CO emissions, insufficient tests were performed to provide a reasonable evaluation of UGA distribution effects.

Furnace particulate emissions were measured during 13 of the characterization tests. Mass loadings varied from 1.20 to 4.55 gr/dscf. The highest particulate emissions were recorded for normal and high loads. A cursory evaluation of the data failed to show any apparent correlations between combustion air distribution and furnace particulate emissions or between the level of particulate emission and the amount of particulate combustibles as measured by "loss on ignition."

INTERPRETATION OF RESULTS

The characterization tests were completed on January 24, 1989. Performance tests were begun on February 13 and completed on February 28, 1989. The interpretation of the characterization tests and subsequent performance tests has been greatly influenced by the experience during the performance tests.

A combustion test plan was developed for the performance tests on Unit 11. This test plan was based on the characterization tests from Unit 11 and some combustion tests CE performed on Unit 12 in the weeks prior to the start of the performance tests. The initial performance test plan called for tests at three average steam load conditions: low - 185,000 lb/hr, normal - 230,000 lb/hr, and high - 245,000 lb/hr. However, Unit 11 could not be run at the planned test conditions while achieving the good combustion conditions anticipated from the previous characterization tests and tests on Unit 12. During initial tests the approach was to set the desired steam load and combustion air conditions, and attempt by careful operation to achieve low CO emissions. This approach was not successful. The approach was subsequently changed to setting the desired combustion air conditions and then limiting the load to the range where good combustion could be achieved.

The inability to operate at high loads while achieving low CO is believed to result from two causes: a mechanical problem with the induced draft (ID) fan on Unit 11 (which does not allow the unit to achieve design combustion gas flow rates) and wet RDF. Heavy rains were experienced during most of the period during which the performance tests were run. This resulted in unusually wet RDF. The major effect of wet RDF is to increase the amount of RDF needed to meet a given load condition. It also increases the concentration of water vapor in the combustor flue gas. The combustion gas flow rate is primarily limited by the ID fan flow capacity. Wet RDF results in higher concentrations of water vapor and decreases the amount of excess air which can be accommodated at high loads.

The demand for combustion air varies with the RDF feed rate and with variations in the RDF properties which affect its rate of burning. Each combustor must be operated with an appropriate level of excess air or combustion air demand will exceed combustion air supply; i.e., temporary increases in RDF burn rates will deplete excess air within the combustor leading to increased CO emissions and increased emission of unburned organics.

Figure 6 shows test average CO concentrations versus average spray dryer inlet O_2 concentration for low, normal, and high load test conditions. The high CO emissions (>350 ppm) for the high and normal load tests at 6 and 7 percent O_2 are probably the result of insufficient excess air margins in the combustor. The variation in CO emissions from normal and low load tests at O_2 concentrations between 8 and 11 percent is postulated primarily to be a result of mixing effects. Above 12 percent, poor mixing and flame quenching are probably the dominant reason for CO emission trends.

Figure 7 illustrates the relationship between steam flow, CO, and O_2, during tests CT-18 and CT-19 (Series E). The average steam flow rate and O_2 concentration for CT-18 was 231,000 lb/hr and 9 percent, respectively. The average steam flow rate and O_2 concentration for CT-19 was 230,000 lb/hr and 7 percent. Data from both tests illustrate the inverse relationship between O_2 and CO and between steam flow and O_2. Increased steam flow corresponds to increased RDF feed rate and increased O_2 demand.

While the differences in CO emission between test CT-18 and CT 19 can be interpreted as a result of changes in undergrate air distribution, the differences may also be the result of lower excess air margins. It is likely that the increased CO emission results both from reduced excess air margins and mixing effects caused by changes in the undergrate air distribution.

Good combustion conditions are illustrated in Figure 7 which shows the relationship between steam flow, CO concentration, and O_2 concentration for performance test PT-10. During PT-10 the average steam flow was 192,000 lb/hr, the average O_2 concentration was 9.2 percent, and the average CO concentration was 85 ppm. The O_2 margin during this test was sufficient to account for variations in O_2 demand.

CONCLUSIONS

The Mid-Conn field tests have provided useful information concerning the effects of operating conditions on emissions. While final conclusions must be based on a more complete evaluation of the test data including the results of the organic, PM, and trace metal measurements, two tentative conclusions are appropriate:

1. Good combustion as measured by CO emission is highly sensitive to load, excess air levels, and distribution/mixing of combustion air.
2. The combustion conditions in spreader stoker RDF combustors can be controlled to provide average CO emissions below 100 ppm.

ACKNOWLEDGEMENTS

The cooperation of the Connecticut Resource Recovery Authority and Combusting Engineering Inc. are gratefully acknowledged. The assistance of R. Michael Hartman of Combustion Engineering who coordinated the tests with plant personnel is especally appreciated.

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APPENDIX

Unit Conversion Factors

English Engineering Unit	Multiplied By	Metric Unit
in.	2.540×10^{-2}	m
klb/hr*	4.536×10^{2}	kg/hr
lb/hr	4.536×10^{-1}	kg/hr
gr/dscf	2.288×10^{3}	mg/dscm
tons/hr	2.268×10^{2}	tonne/h
°F-32	5.555×10^{-1}	°C

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*Note that KPH in Figure 7 represents klb/hr.

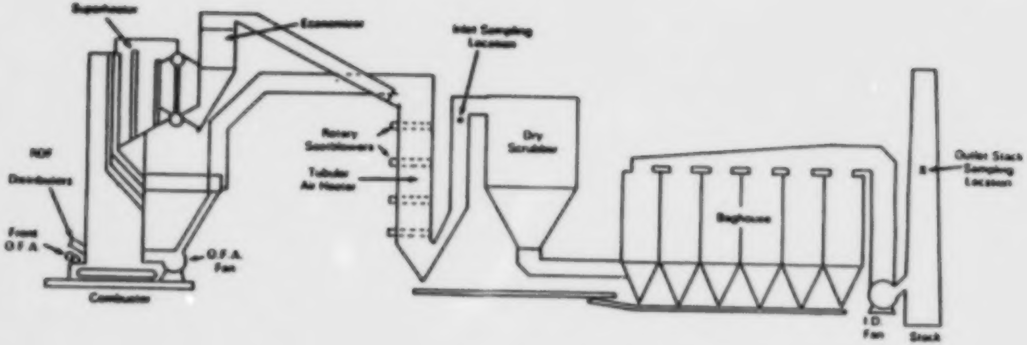
Superheater
Economizer
Inlet Sampling
Location
RDF
Distributors
Front
O.F.A.
O.F.A.
Fan
Combustor
Rotary
Sootblowers
Tubular
Air Heater
Dry
Scrubber
Baghouse
Outlet Stack
Sampling
Location
I.D.
Fan
Stack

Figure 1. Schematic flow diagram for Unit 11 of the Mid-Connecticut Resource Recovery Facility with sampling locations.

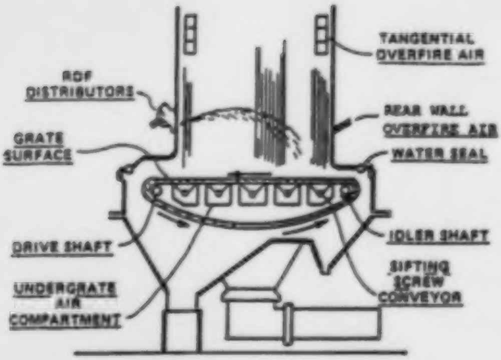
TANGENTIAL OVERFIRE AIR
RDF DISTRIBUTORS
REAR WALL OVERFIRE AIR
GRATE SURFACE
WATER SEAL
IDLER SHAFT
DRIVE SHAFT
SIFTING SCREW CONVEYOR
UNDERGRATE AIR COMPARTMENT

Figure 2. Combustion Engineering RDF spreader-stoker at Mid-Conn.

SERIES A, FULL LOAD

CT	13	6	9	8	13A	2	7	12	4	12A
TOFA	1	2	2	2	2	3	3	3	3	3
-TOP	-	C	C	C	-	C	C	4R	C	4R
-MID	-	4L	4L	4L	C	4L	4L	C	4L	C
-BTM	C	-	-	-	C	4R	4R	C	4R	C
RWA	35	-	15	35	35	-	-	15	35	35
UGA	II	II	II	II	II	II	II	II	II	II
STEAM	232	226	231	216	226	221	229	219	227	231
O2	7	8	8	10	10	9	9	11	8	8
CO	455	141	253	73	247	NA	229	158	NA	52
TEMP	1901	1848	1834	1837	1802	1913	1847	1765	1893	1848

NOTES

CT = Characterization test no.
TOFA = No. of levels of overfire air
TOP = Angular orientation of top row of TOFA
MID = Angular orientation of middle row of TOFA
BTM = Angular orientation of bottom row of TOFA
Temp = Calculated average combustion gas temp, °F
Steam = Boiler steam flow rate, 1000 lb/hr
O2 = Test average O2 concentration at SD inlet, ppm
CO = Test average CO concentration at SD inlet, ppm
RWA = Rear wall air plenum pressure, in. H2O

UGA = Undergrate air damper positions

UGA

I	II	III	IV
15	15	15	15
90	90	90	90
90	90	90	90
90	90	45	45
45	30	30	15

Figure 3. Combustion conditions for full load characterization tests.

SERIES B, LOW LOAD

CT	11A	11B	11C	5	16	20B	3	10A	10B
TOFA	0	0	0	1	2	2	3	3	3
-TOP				C	-		C	C	C
-MID				-	2L		4L	4L	4L
-BTM				-	C		4R	4R	4R
RWA	-	15	35	-	35	35	-	-	-
UGA	II	II	II	II	II	II	II	II	II
STEAM	151	149	161	153	143	180	150	153	192
O2	12	13	14	12	16	12	17	17	11
CO	952	768	461	151	222	343	NA	407	257
TEMP	1912	1887	1794	1713	1675	1812	1673	1684	1853

NOTES

CT = Characterization test no.
TOFA = No. of levels of overfire air
TOP = Angular orientation of top row of TOFA
MID = Angular orientation of middle row of TOFA
BTM = Angular orientation of bottom row of TOFA
Temp = Calulated average combustion gas temp, °F
Steam = Boiler steam flow rate, 1000 lb/hr
O2 = Test average O2 concentration at SD inlet, ppm
CO = Test average CO concentration at SD inlet, ppm
RWA = Rearwall air plenum pressure, in. H2O

UGA = Undergrate air damper positions

UGA			
I	II	III	IV
15	15	15	15
90	90	90	90
90	90	90	90
90	90	45	45
45	30	30	15

Figure 4. Combustion conditions for low load characterization tests.

SERIES C, HIGH LOAD

CT	15A	15B	14	20A
TOFA	3	3	3	3
-TOP	C	C	C	2L
-MID	C	C	C	1L
-BTM	C	C	C	C
RWA	-	-	35	35
UGA	II	II	II	II
STEAM	246	240	244	245
O2	6	6	7	7
CO	381	392	406	426
TEMP	1925	1901	1891	1880

SERIES E, UGA
DISTRIBUTION

17A	17B	18	19
3	3	3	3
2L	2L	2L	2L
1L	1L	1L	1L
C	C	C	C
35	35	35	35
III	IV	II	IV
207	222	231	230
11	10	9	7
158	106	179	385
1731	1729	1819	1888

NOTES

CT = Characterization test no.
TOFA = No. of levels of overfire air
TOP = Angular orientation of top row of TOFA
MID = Angular orientation of middle row of TOFA
BTM = Angular orientation of bottom row of TOFA
Temp = Calculated average combustion gas temp, °F
Steam = Boiler steam flow rate, 1000 lb/hr
O2 = Test average O2 concentration at SD inlet, ppm
CO = Test average CO concentration at SD inlet, ppm
RWA = Rear wall air plenum pressure, in. H2O

UGA = Undergrate air damper positio

UGA

I	II	III	IV
15	15	15	15
90	90	90	90
90	90	90	90
90	90	45	45
45	30	30	15

Figure 5. Combustion conditions for high load and UGA distribution characterization tests.

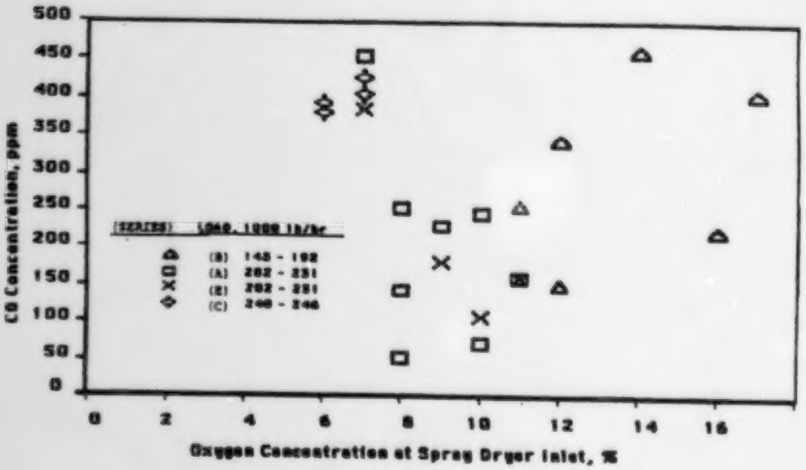

Figure 6. Test average CO concentrations versus oxygen concentration at the spray dryer inlet.

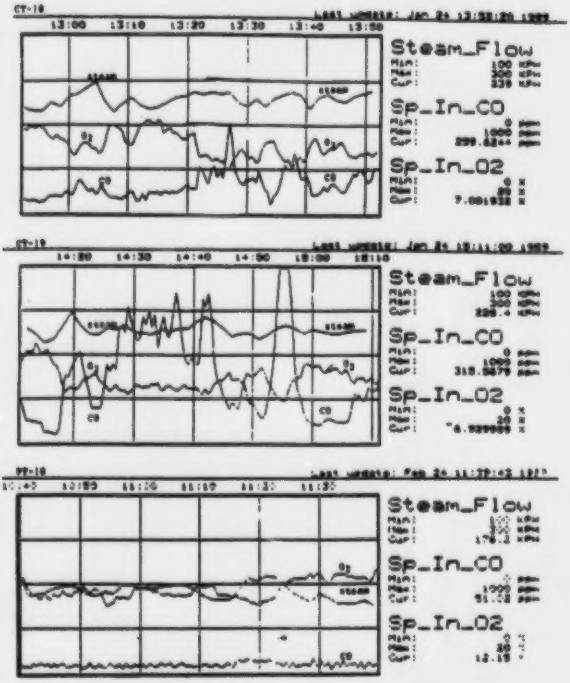

Figure 7. Illustration of relationship between steam flow, CO concentration, and O_2 concentration.

SORTED INCINERATOR SLAG USED AS FILL MATERIAL

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ABSTRACT

In Sweden, 400 000 tons of incinerator slag is produced per year. So far only a small portion is utilized. To increase the value of the slag, a special product is produced by screening and magnetic separation. This product is called ash-gravel and its proportion of the total slag amount is 70 per cent.

The paper presents tests made in laboratory and field to clarify the properties of ash-gravel as a fill material to substitute natural sand and gravel. Special emphasis has been directed to compaction properties, effect of slag age and risk of crushing due to compaction.

BACKGROUND

In Sweden today, incinerator slag is produced at 27 plants with a total production of more than 400,000 tons/year. According to the Swedish Association of Public Cleaning and Solid Waste Management (1), there is a potential for utilizing some 250,000 tons/year of this type of slag (botto ash). However, the present level of utilization is far lower than this. One of the main reasons being that there are no official regulations or guidelines for evaluating the risk of environmental disturbance and thus determining in advance whether approval will be granted in a particular project. Owing to the lack of guidelines, there are always requests by authorities for environmental scenarios concerning each individual project in order to judge the environmental impact. To achieve a more regular use of slag, a large-scale project is now in progress in Sweden, which includes the following aspects: rules for obtaining official permits, sampling and analysis methods, evaluation of environmental impact and areas of utilization.

The main purpose of the project is to clarify the potential use of sorted incinerator slag. In this paper, tests on using such slag as filling material are described. This has been done through compaction tests in the laboratory and in the field, using slag of different ages. The field tests have mainly been carried out at the site currently used for slag disposal in Malmeo. The study comprises factors such as influence of the age of the slag, choice of compaction machines and performance, changes in slag properties due to heavy compaction and influence of weather conditions (2).

ASH HANDLING AT THE PLANT IN MALMÖ

Combustion in the Malmö plant (SYSAV) takes place in a grate furnace. The slag is removed from the bottom of the furnace and is slaked in a water bath. The slag constitutes about 28% of the weight of the refuse prior to combustion. The Malmö incineration plant produces some 60,000 tons of slag and about 4,000 tons of flue gas cleaning products every year.

The view in Sweden, as in several other European countries (3), is that the incinerator slag must be screened before it can be used. For this reason, SYSAV introduced a sorting plant for slag in 1984. Sorting makes use of screens as well as drum magnets placed above the conveyor belt. The fractions obtained after sorting are shown in Figure 1.

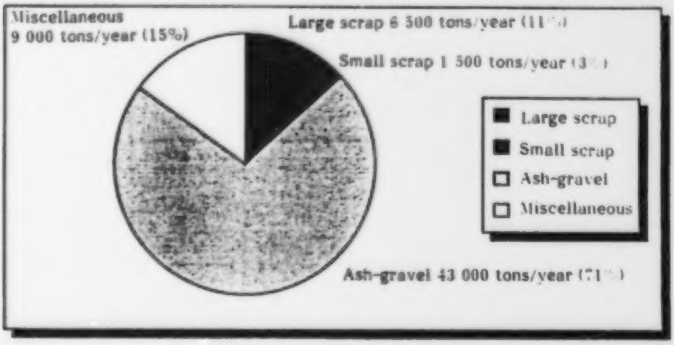


Figure 1. Proportion of different fractions of a sorted incinerator ash-gravel from Malmö.

At present, the "miscellaneous" fraction is deposited, but could probably be used under controlled conditions as general fill material. The large scrap is sold, while the small scrap sometimes is sold sometimes deposited. Remaining material, given the name ash-gravel, constitutes 70% of the total and has been used in the tests described here. The material resembles a relatively well-sorted gravel-sand material, but has a relatively high content of fines (over 10% with a fraction of max 0.074 mm). The fresh ash-gravel has proved to be somewhat coarser than the aged ash-gravel. Particle size distribution curves for these materials are shown in Figure 2.

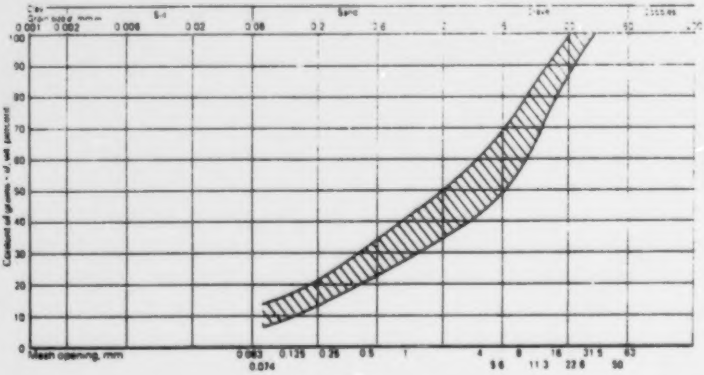


Figure 2. Particle size distribution of ash-gravel from Malmö incinerator plant.

TEST PROGRAM

In order to determine the effect of different types of compaction roller and the significance of the age of the ash-gravel in compaction, four test embankments have been built. Since the maximum layer thickness depends on the weight of the roller and two rollers were used, the embankments had a different overall height. Half the length of each test embankment was built up from fresh ash-gravel (max 3 days old) and the other half from aged ash-gravel (about 3 years old). At an early stage of the tests, it was found the fresh ash-gravel was not performing satisfactorily. Build-up of most of the embankments with fresh material was therefore stopped before the full height was reached. The general design of the test embankments is shown in Figure 3.

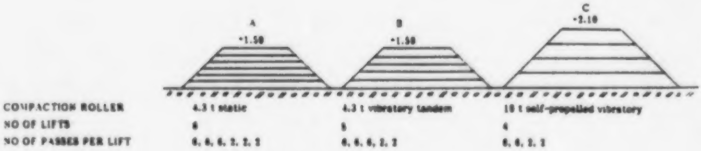


Figure 3 Build-up of the test embankments.

Two types of rollers were used; a 10-ton self-propelled vibrating roller (Dynapac CA 25D), and a 4.3-ton vibratory tandem roller (Dynapac CC14). In one of the sub-surfaces (embankment A) the vibrating function was disconnected, so that the roller acted as a static roller. The layer thickness was 0.55 m for the 10- ton roller and 0.25-0.30 m for the 4.3-ton roller.

Before the first layer was applied, the subgrade was compacted consisting of slag laid at an earlier stage. The same roller was then used as later in the particular embankment. The number of roller passes was 6 in all cases.

CRUSHING OF PARTICLES DUE TO COMPACTION

An important question when ash-gravel is to be used as a substitute for sand and gravel is the risk of crushing of the particles during laying and compaction, in addition to its long-term durability.

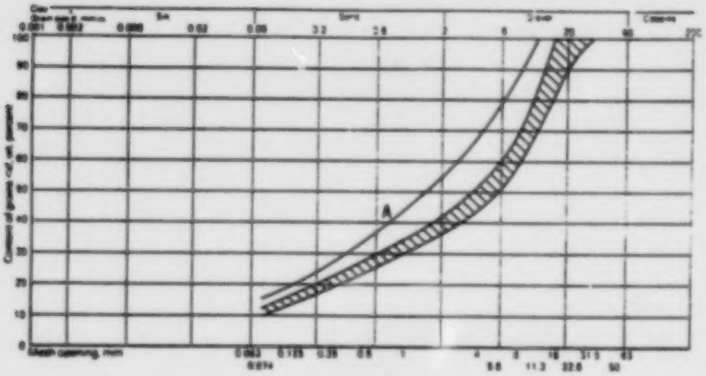
The change in the particle size distribution resulting from crushing during compaction has been studied both in field experiments and in the laboratory. In the field experiments, samples were taken by excavating test pits in all test embankments after compaction had been completed. In general, it was found that a certain variation in the particle size distribution occurs, which can be attributed to the natural variations in the ash-gravel. It was found that the number of roller passes, choice of static or dynamic compaction, roller weight and age of the ash-gravel have only a marginal effect on the degree of crushing (Table 1).

TABLE 1. Particle size of ash-gravel after compaction in the field (2).

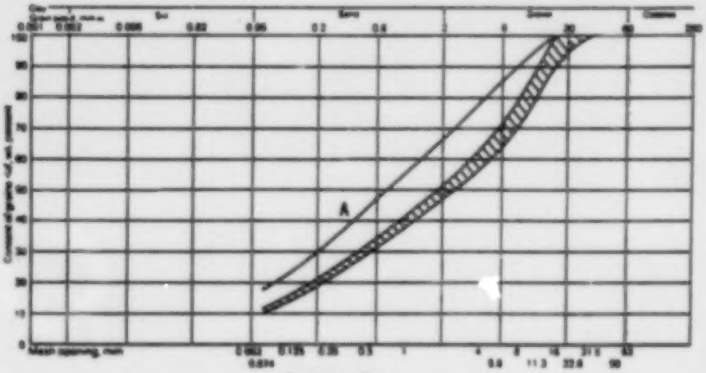
	No. of passes	Aged, %		Fresh, %	
		<0.072 mm	>2 mm	>0.075 mm	>2 mm
10-ton vibratingting roller	6	9	50	11	57
	2	7	58	8	51
4.3-ton vibrating roller	6	9	57	9	65
	2	11	53	-	-
4.3-ton static roller	6	11	54	15	54
	2	9	52	-	-

The laboratory tests were performed on ash-gravel with a somewhat different particle size distribution to that in the field experiments, because these tests were performed at an earlier stage. Figure 4 shows that the fresh ash-gravel then was somewhat coarser than the aged ash-gravel before compaction. The content of particles larger than 4 mm was 55% for fresh and 43% for aged ash-gravel. In both cases, a certain crushing was obtained with modified Proctor. The aged ash-gravel broke down to a greater extent than the fresh ash-gravel, indicated by the content of fines increasing to 17% in the aged material as against 13% in the fresh material. The proportion of particles larger than 2 mm fell from about 40% to about 20% for the aged ash-gravel and from about 50% to about 30% for the fresh ash-gravel.

A comparison between field and laboratory experiments shows that compaction in the laboratory with heavy compaction results in greater crushing than that actually obtained in the field. The same conclusion has earlier been drawn in the case of coal slag. A preliminary recommendation is therefore to study the degree of crushing in the field through Standard Proctor compaction or vivratory compaction in the laboratory.



2 months old



1 year old

Figure 4. Particle size distribution of ash- gravel before and after modified Proctor. "A" indicates the particle size distribution after compaction.

WATER CONTENT

When conducting the field tests, samples were taken for determining water content and particle size distribution. The water content determined in ash-gravel before compaction is shown in Table 2.

TABLE 2. Water content of ash-gravel before laying in the field.

	No.	Mean	Standard deviation
Aged ash-gravel	13	16.4%	1.3%
Fresh ash-gravel	5	22.9%	1.5%

Samples were taken by excavation to various depths after the test embankments had been built up to their full height. The water content was then as in Table 3.

TABLE 3. Water content after compaction (2)

	No.	Mean	Standard deviation
Aged ash-gravel	15	20.1%	3.3%
Fresh ash-gravel	10	27.6%	5.2%

A comparison between ash-gravel before and after compaction shows that in both aged and fresh ash-gravel the water content increased during compaction. This may be explained by precipitation and the fact that water was pumped up from the underlying fill.

During compaction, the increased water content of the fresh ash-gravel led to a loss of bearing capacity in the fill. Figure 5 shows a ground failure under the 4.3-ton roller. This phenomenon can also be obtained in natural fine-grained soils when the water content exceeds the optimal level by 3-5 per cent units according to modified Proctor compaction.

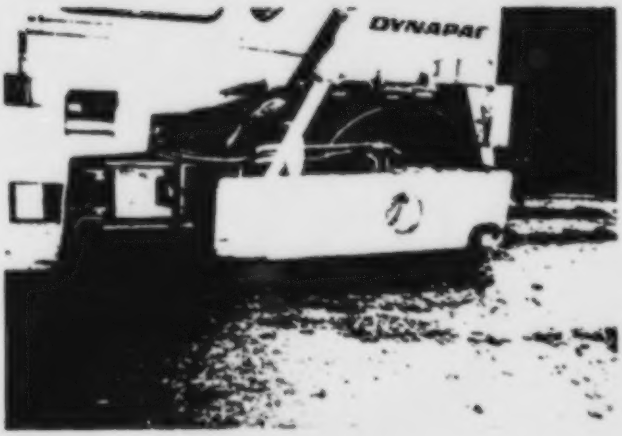


Figure 5. Ground failure in fresh ash-gravel beneath a 4.3-ton roller.

Preliminary results from CBR tests show that high values are obtained and that maximum stability is reached a few per cent units below the optimal water content (4). After water saturation of compacted samples, stability increased further. No great differences were observed between 2 months old and aged ash-gravel. However, it should be noted that fresher ash-gravel would probably have led to even poorer stability.

EFFECT OF COMPACTION

In the laboratory, compaction tests were performed on fresh ash-gravel, ash- gravel 1 month old and on an aged ash-gravel (over 1 year old). As shown in Figure 6, a clear compaction curve was obtained with a well-defined peak only for the aged material. However, the maximum recorded values are of the same order of size for the different ages of ash-gravel, i.e. 1.79 to 1.82 t/m^3. These values are considerably higher than in earlier laboratory tests which gave densities of about 1.69 t/m^3 (4).

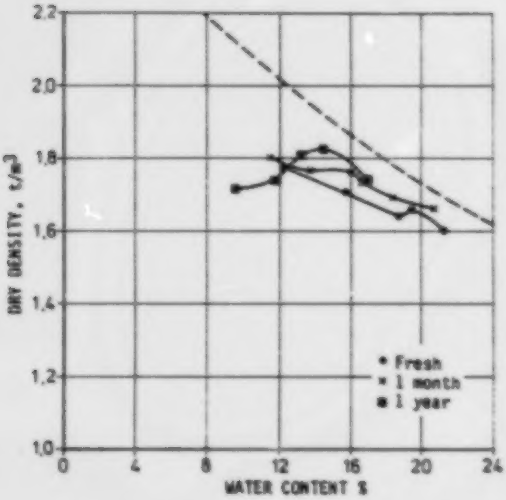


Figure 6. Compaction curves (modified Proctor) of sorted incinerated gravel-ash of three different ages (2).

In connection with test pit excavation in the finally compacted test embankments, the density was also determined by water balloon. The measured densities were higher for old ash-gravel than for fresh material under similar conditions. Increasing the number of roller passes from 2 to 6 raised the dry density of the aged ash-gravel by about 5-10%. In the case of the fresh ash-gravel, the number of tests was too small to evaluate the effect of increased compaction energy.

During compaction, it was found that the aged ash-gravel was very easily compacted, while the fresh rebounded upon each roller pass. At the same time, water was forced up from the fill consisting of fresh ash-gravel. The recorded water contents were greater in the fresh ash-gravel, which probably resulted in the lower degree of compaction and the resilient properties.

With a maximum dry density of 1.82 t/m3, degrees of compaction as shown in Table 4 were obtained in field tests. It was found that a 90% degree of compaction can be obtained for aged ash-gravel, especially with a 10-ton roller. The uncertainty lies in obtaining representative samples in the field. The scatter indicates that the ash-gravel was inhomogeneous or maybe,that the sample volume was too small using the standardized water balloon.

In the case of the fresh ash-gravel, a maximum degree of compaction of 85% was obtained.

TABLE 4. Degrees of compaction obtained in compacted aged slag in the field (2).

Roller type	No. of passes	Degree of compaction, %
10-ton vibrating	6	91-94
	2	90
4.3 ton vibrating	6	82-90
	2	78
4.3 ton static	6	82-89
	2	83

Chemical reactions may lead to a certain self-binding in the slag. Determination of the E-modulus in Swedish slags(5) shows both that sorted ash-gravel has a higher firmness than fresh slag and that a tangible self-binding may be obtained in aged slag (Figure 7). However, both firmness and self-binding were highly dependent on the moisture content. This sensitivity to water may be one of the reasons for the differing information in the literature on self-binding in slag fillings.

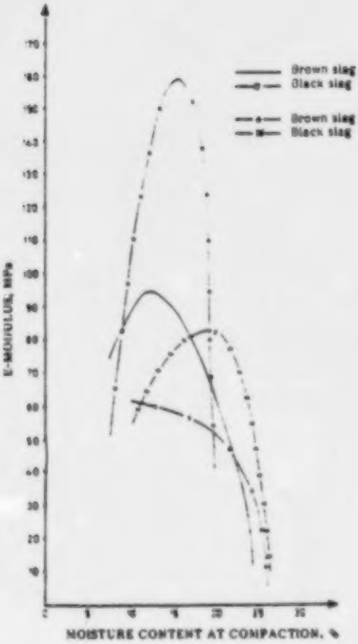


Figure 7. E-modulus according to the SEB method for brown aged slag and black fresh slag respectively from Lövsta without sorting. Loading applied both immediately after heavy compaction and after 28 day's storage in moist environment (5)

USE IN AN ABUTMENT FILL

After the test embankments had been examined, compaction was performed adjacent to test compaction areas around a bridge abutment. The embankment was compacted with a 10-ton vibrating roller, 0.55 m layer thickness and 6 roller passes, i.e. in the same way as test embankment C. The total depth of the fill was about 3.0 m.

In order to determine the firmness, pressometer tests were performed at three depths beneath the upper surface, corresponding to 0.9, 1.8 and 2.7 m. Owing to the limited area of the fill, only one successful hole was made. In this central hole, pressometer moduli of 32.8, 38.9 and 29.7 MPa were obtained. These values indicate that the fill is very firm. In order to achieve such a high firmness, chemical reactions must have taken place in the fill after compaction, resulting in hardening.

A comparison with other fills shows that the firmness of the ash-gravel corresponds to that obtained in compacted rock fills and is therefore greater than that obtained in a compacted sand fill. However, it should be noted that experience so far is very limited in the case of ash-gravel fills. The pressometer tests show that it should be possible to obtained a very firm fill when using slag from incinerator plants.

CONCLUSIONS AND FURTHER INVESTIGATIONS

Experience in Sweden, as in other European countries (6), shows that it is important to sort the slag, thereby eliminating uncombusted and metallic materials.

Compaction tests performed in the field have shown that good performance is obtained with a sorted slag stored for at least 1 year. Compaction can be carried out with a conventional method standardized for Swedish conditions. Degrees of compaction exceeding 90% according to modified Proctor can be achieved, especially with a heavy vibrating roller (say 10 tons). Fresh ash-gravel has proved to be unsuitable, even though the laboratory test did not indicated any difference in comparison with aged ash-gravel. In the field, a resilience was obtained during compaction and water was pressed out of the fill. The excess water led to a ground failure beneath one of the rollers during compaction (see Figure 5). The reason why fresh ash-gravel does not perform as well as aged ash-gravel is principally believed to be the higher water content. Also oxidation during aging may change the properties of the ash-gravel. The compaction tests described in Figure 6 show that the material changes in character over time.

Further investigations will be conducted partly in order to clarify the necessary properties for using ash-gravel in advanced applications. At present, an industrial road 600 metres long is being constructed of incinerator ash-gravel. Results achieved so far indicate that the sorted incinerator slag has very good properties. It also seems to be suitable as filling material in structural fills.

The environmental considerations are very important when utilizing incinerator ash. To evaluate the impact, leachate collection systems have been placed underneath two road embankments. The system is designed so that leachate water is collected both under the asphalted area and under the slopes covered with grass. The content of heavy metals in the leachate water is compared to the results from leaching performed in the laboratory (Kullberg et al 1989).

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ENGINEERING MANAGEMENT OF MSW ASHES:

FIELD EMPIRICAL OBSERVATIONS OF CONCRETE-LIKE CHARACTERISTICS

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ABSTRACT

The principles of proper Site Management, including adding the optimum water of solubilization and attaining optimal compaction, were applied to a series of Test Patches subject to Portland Cement and lime [CaO] in-situ chemical treatment. Mechanically processed and compacted Municipal Solid Waste [MSW] ash achieved in-place compacted dry density of 113.31 pcf and compacted wet density of 134.82 pcf. Such results achieve 48% volumetric reduction over typical ash landfill practise and save from 89% to 93% of the land usage, as compared to MSW land disposal. Empirical in-place field testing at a Field Demonstration Program yielded permeability coefficients between 10 X EXP-6 cm/sec to 10 X EXP-9 cm/sec, after 14-28 days curing; determined by laboratory and field measurements. The results of prior laboratory investigations were confirmed in terms of Portland Cement and CaO treatment of MSW ash. Such values reflect the benefit of in-situ chemical treatment to achieve liner-like permeabilties. The addition of CaO achieved an order of magnitude reduction in permeability; suggesting the pozzolanic behavior of MSW ashes. Assuming a worst-case 10% Portland Cement addition for In-Situ Chemical Treatment, applied by an outside specialty contractor, savings of $ 200,000 to $ 450,000 per acre could be realized; compared to single, double, synthetic and clay liners, currently under review for MSW Ash Monofill implementation. The pending legislation fails to consider such properties and pozzolanic-like behavior and to recognize the application of engineering management principles.

INTRODUCTION

The growth of municipal solid waste-to-energy facilities in the US is due primarilly to the closure of Municipal Solid Waste [MSW] landfills, the desire to recover energy from MSW, the need to manage waste in a sanitary fashion and a crucial need for conserving land use. Recently regulators haved focused on the environmental issues and evaluation of possible hazards associated with waste incineration. One of the major environmental issues surrounding the refuse-to-energy industry is the disposal of MSW ash residue. In 1988, this MSW ash will amount to between 2.5-4 million tons and its annual generation rate is expected to increase to 17 million tons by the year 2000 (1).

MSW ASH HANDLING

A typical Flue Gas Cleaning [FGC] system consists of reacting the incinerator flue gas with lime (usually in an absorption vessel) followed by particulate removal (baghouse or electrostatic preciptiator). While these systems are designed to remove Acid Gases and particulates, they increase the waste generation rate due to the product of reaction products and unreacted lime. The use of FGC systems do not increase the ashes' organic content, since only CaO is added in the dry absorption stage. The FGC waste is composed of Fly Ash [FA] and dry Scrubber Residue [SR]) reaction products. The ash generated from Mass Burn MSW systems is composed primarily of BA (85-90 weight%) and FA (10-15 weight%). Equipped with an AGC system, the combination of BA, FA and SR amounts to 1/4 to 1/3 of the weight of the MSW feedstock.

As depicted in Figure 1, the Fly Ash/Scrubber Residue may be combined with the Bottom Ash, which has been quenched, hence the resultant blend will contain water. The mixture of ash and FGC wastes is conveyed to a disposal site. This Combined Ash [CA] may be thixotropic and contains considerable lime. The solids content should be maintained to ensure optimal transportability characteristics i.e. (a) prevention of fugitive dusting; (b) elimination of thixotropic behavior; (c) prevention of pre-mature pozzolanic reaction. Typically, the transportation solids content could range from 75-85 % to satisfy these criteria.

ASH MANAGEMENT BY FEDERAL LEGISLATION

During the 1988 Congressional Session, three proposed pieces of legislation could impact the present regulation of MSW ash: (a) Senate Bill [S. 1894, S. 2458]; (b) Florio Bill [H.R. 4902]; (c) Luken Bill [H.R. 4357]. Rather than discuss these proposals in detail, Kochieisen (2) has provided a comprehensive comparative analysis. Common points of this analysis reflect regulatory approaches which justify commentary. These common points are listed, as follows:

o Minimum Requirements of Ash Disposal Monofills

o Design Requirements

o Test for Ash Toxicity

Thus the regulatory proposals reflect just the issues concerned with the ash testing and design of ash landfills. These proposals fail to discuss the engineering management of the MSW ash and to consider available empirical data.

COMPOSITION

Table 1 compares the mineral composition of MSW Combined Ash [CA]; (i.e. the combination of BA, FA and SR) to Portland Cement. Similar compositions have been reported for BA and FA for older MSW systems, without Flue Gas Cleaning, i.e. no lime-based dry scrubber; however, the CaO content was reported as approximately an order of magnitude lower (3). The greater lime content is due to the dry scrubber stoichiometries (exceeding 1.5) associated with MSW Flue Gas Cleaning (4).

Lime (CaO) in the presence of Silica (SiO_2), Alumina (Al_2O_3), and Calcium Sulfate ($CaSO_4$) form sulfo-alumina hydrates (ettringites) and calcium silica hydrate (tobermorite) as represented by the following:

Ettringite: $3CaO.Al_2O_3.3CaSO_4.28\text{-}32H_2O$
Tobermorite: $CaO.SiO_2.nH_2O$

These two mineral end products form a concrete-like material or pozzolan; a form of solidification that encapsulates the waste in a monolithic solid of high structural integrity. This reaction yields a solid, monolithic mass with sufficient structural integrity to be handled, transported and disposed. The relatively low permeability of the final product reduces the leachable surface area and minimizes the solubility of heavy metal components of the waste (5,6,7).

ENGINEERING CONSIDERATIONS MSW ASH SITE DISPOSAL MANAGEMENT

MSW Ash possesses the potential for pozzolanic behavior due to the presence of high free lime content. Recognizing this potential pozzolanic reaction is critical to incorporate into residue disposal site management practises. Previous laboratory investigations on MSW ash and analagous pozzolanic material have reported on the soil-like high density and low permeability properties of MSW Ash (8,9). Prior studies have demonstrated that MSW ash [BA, FA, SR] monofill disposal reflects a Toxicity Index Ratio of 0.04 - 0.06 to raw MSW landfilling (10). Such conclusions, developed solely from laboratory and theoretical considerations, require field empirical verification to demonstrate the non-hazardous nature of MSW ash monofill disposal and to capitalize on the high densities and liner-like permeabilities of MSW residues. The empirical results derived from field demonstration activities, reported previously (9) provides (a) confirmation of in-place geo-technical properties to laboratory results and (b) non-hazardous nature of MSW ash monofill leachate and runoff.

LABORATORY AND FIELD PROGRAMS GEO-TECHNICAL RESULTS

The geo-technical results reported in this paper reflect the co-author's previous derivation of such technology in terms of a graphical-mathematical approach to FGD sludge and power plant ash (11). Inherent to this approach are (a) achieving optimal moisture content and (b) solubilization of free, available lime. Achieving these objectives assures attaining high compacted density and pozzolanic behavior of residues.

PARTICLE SIZE DISTRIBUTION AND COMPACTED DENSITIES

Although a comprehensive laboratory program (including investigations using kiln dust, Fly Ash from coal-fired plants, and gypsum) was conducted, only results comparable to the field program are reported. The MSW ashes investigated in the laboratory program were generated from Resource Recovery Facility I equipped with a lime-based FGC System. Unlike Facility I, the ash studied in the field program was generated at an older Facility II, not equipped with a lime-based FGC System.

Figure 2 depicts the Particle Size Distribution of raw BA and CA from Facility I. BA (I) reflects a granular material of mostly gravel consistency. CA (I) represents a finer material, having almost equal gravel and sand consistencies. These ashes were compacted in accordance with the modified Proctor Test [ASTM D-1557]. The results of this testing are shown on Figures 3 and 4. Figures 5 and 6 depict the Particle Size Distribution and Proctor Testing, respectively, of CA (II). This material's distribution, twice as much gravel consistency as sand, is bracketed by BA (I) and CA (I) distributions and reflects proprietary mechanical processing.

All ashes were compacted at a non-optimal moisture content to facilitate achieving the desired % Water of Solubilization. Ashes from facility I represent modified Proctor Compactions; while CA (II) compacted densities and moisture contents were measured in the field; using a Nuclear-Density meter. The reported moisture content results include a field correcting factor achieving an error of less than +/_ 1%. Table 2 provides a comparison of Particle Size Distribution and Compacted Dry Density for all three ashes. The compactive effort of modified Proctor test is suitable to simulate heavy field equipment used in airfield construction (12). The compacted density of CA (II) was achieved by compaction with a 20 ton vibratory roller. Therefore, the compactive effort in the laboratory and field are comparable. The highest density, achieved for CA (II), reflects its proprietary mechanical processing.

Table 2 also reports the in-place density and moisture content of the CA (II) before and after compaction. The ash's wet density of 134.82 pounds per cubic foot (pcf) reflects the unit weight of the solids and water as compacted in-place. The ash's dry density of 113.31 pcf reflects its degree of saturation as related to the Proctor compaction curve. In-place compaction and solubilization water addition represent the application of engineering principles to achieve enhanced geo-technical properties.

VOLUMETRIC REDUCTION

The application of an engineering approach to ash disposal attains signficantly higher densities and lower volumetric requirements. The significance of obtaining compacted dry density of 113.31 pcf, for this mechanically processed and well compacted ash, can be realized by considering typical ash and raw MSW densities.

Typically, MSW ash landfills have been sized on the basis of a wet density = 74 lb/cf for % water content = 15 - 25%. By assuming 20% water, the typical dry density for landfill sizing = 59.2 pcf. The following derivation represents volumetric reduction in terms of initial and final densities:

$$\text{Change in Volume} = (Vo - Vf)/Vo \qquad (1.0)$$

Where Vo = initial volume; Vf = final volume; Vo > Vf

Since DDo = initial dry density = Wo/Vo (where Wo = initial weight) and DDf = final dry density = Wf/Vf (where Wf = final weight) and Wo = Wf (i.e. dry solids remain the same), then

$$\%\ \text{volumetric reduction} = [1 - (DDo/DDf)] \times 100\% \qquad (2.0)$$

Both mechanical processing and field compaction sigificantly reduces volumetric ash monofill requirements. By applying equation (2.0); achieving a 48% volumetric reduction over typical ash landfill practises can be calculated. MSW landfills are typically sized on an 'as-delivered' density of 52 - 60 pcf (compacted with sheepefoot roller) and of 37 pcf (without compaction). Taking into account the approximate 25 % weight reduction, achieved from waste-to-energy systems, and applying equation (2.0) with a compacted CA (II) wet density of 134.82 pcf, MSW ash (processed and compacted) monofill's volumetric reductions would be 89-90 % and 93 %, respectively, of landfilling MSW.

Field compaction amounted to an 8% volumetric reduction; while the remaining 40% volumetric reduction was due to proprietary mechanical processing, yielding an improved solids distribution. The shape of a particle size distribution affects the nesting of the different sized particles. A material, having uniform distribution of large, medium, and fine sized particles, should permit for more solids occupancy and less voids occupancy for a given volume of material (13). Given a material with a higher solids content, per unit volume, a greater density will be achieved.

FIELD PROGRAM DESCRIPTION

A field program, designed to demonstrate the viability of low-cost in-situ chemical treatment achieving liner-like permeabilities, was initiated at Facility II. Ash from this a faciltity, not equipped with a FGC system, represented an opportunity to demonstrate the cost-effective methodology of in-situ addition of Portland Cement and of lime [CaO] to non-chemically reactive MSW ash. The Facility II Ash used in the field program was subject to proprietary mechanical processing prior to placement.

TEST PATCH DESCRIPTION

Three (3) Test Patches were formed according to the following dimensions: 20 ft. X 20 ft. (plan area) and 12 inches depth. Chemical additives and water of solubilization were added to a 7 inch depth. In-situ permeameters (of the double-walled type) were installed to a 6 inch depth; providing a margin depth for lateral and vertical leaching. The following denotes respective Test Patch designation and chemical dosage:

Test Patch Designation	Chemical Dosage (% by weight)
No. 1	Portland Cement [PC] - 10.0 %
No. 2	Portland Cement [PC] - 6.6 %
No. 3	Lime [CaO] - 6.6 %

Ashes used in the Laboratory Program were composite sampled from facility I, which is equipped with a FGC system contributing unreacted CaO to CA (I).

PERMEABILITY TESTING

Three different permeability tests were employed to provide sufficient bracketing of in-situ and laboratory results in support of empirical high-quality leachate results demonstrated at the Facility I ash mono-fill (9). In-situ results (28 days field cured) were obtained by employing a modified double-wall permeameter (based on ASTM 3385). Cores, extracted at 28 days field curing, from each test patch were subject to both Triaxial permeability determinations and Falling Head permeability testing (COE EM 1110-2-1906). In addition, field mixtures, of molded material from each test patch, and laboratory mixes, of each test patch chemical dosage, were subject to Falling Head permeability testing. The field and lab mixes are reported at 14 days curing. The laboratory program conducted Falling Head permeability tests of composite samples of ashes from Facility I.

Permeability results of the field and laboratory programs are reported in Table 3.

CHEMICAL (POZZOLANIC-LIKE) REACTIONS

Bottom Ash [BA] should not contain free available CaO, since it is collected upstream of the FGC system. Combined Ash [CA (I)] represents the combination of BA with the separately collected FA and SR. Since the CA (II) does not reflect lime contribution from a FGC system, the BA (I) should represent a similar composition. Table 3 reports that CA (II) + 0% exhibited a laboratory permeability of 1.9 X E-5 cm/sec; practically equivalent to raw BA (I) permeability of 1.8 X E-5 cm/sec. Thus, the 6.6% CaO Test Patch CA (II) permeability results can be compared to Lab Program results for CA (I) with and without CaO addition. These latter ashes were obtained from the Facility I reflecting significant inherent CaO due to high stoichiometry of the FGC system.

The Test Patch Program (Table 3) reports permeability results from 6.4 X E-6 cm/sec to 2.3 X E-8 cm/sec. Since the lowest result may be an anomoly, the CA (II) with CaO exhibits 10 X E-6 cm/sec order of magnitude permeability. This order of magnitude permeability reduction in the presence of free CaO suggests pozzolanic behavior. The raw CA (I) permeability of 5.5 X E-6 cm/sec could reflect the presence of excess lime contributed from the operating FGC system. Upon the addition of lime to CA (I) permeabilities ranging from 8.1 X E-7 cm/sec to 4.2 X E-6 cm/sec were achieved. These permeabilities agree with field measurements. Both sets of results demonstrate an order of magnitude reduction of permeability; suggesting the presence of a lime-based pozzolanic reaction.

EFFECT OF ADDING PORTLAND CEMENT

The variation between in-situ and laboratory permeabilities reflect typical field and laboratory testing differences (14). Adding 10% PC to Test Patch ash CA (II) reduced the permeability by two to four orders of magnitude. The maximum in-situ permeability, measured within Test Patch No. 1, of 2.8 X E-9 cm/sec achieved with 10% PC added to CA (II) compares similarly to a permeability of 1.7 X E-8 cm/sec for BA (I) with 9% PC added. The maximum in-situ permeability, measured within Test Patch No. 2, of 3.53 X E-7 cm/sec agrees with the laboratory permeability of 1.5 X E-7 cm/sec for BA (I) with 6% PC. Thus, the addition of 6-10% Portland Cement added to non-reactive MSW ash attained permeabilities varying from slighly greater to at least an order of magnitude less than the liner requirement of 1 X E-7 cm/sec.

COST SAVINGS

Based on one foot of CA with 10% PC and a compacted density of 100 pcf, one foot of CA with 10% PC should cost $ 3,900/acre; i.e. assuming placement and equipment by landfill operator (8). If an outside specialty contractor is employed, the cost of forming such a liner could increase to approximately $ 50,000. Comparison to single, double, synthetic and clay liners (costing from $ 250,000 to $ 500,000 per acre) indicates that significant cost savings ($ 200,000 to $ 450,000) can be realized by applying in-situ chemical treatment and engineering methodology to MSW ash disposal.

The lower permeabilities, achieved with MSW ash treated with PC, than those, attained with CaO addition, may be due to less reactive CaO. Further testing, over extended curing time, is in progress to determine if lower CaO based permeabilities are achieved.

SUMMARY/CONCLUSIONS

The following summarizes the Laboratory and Field Program results of geo-technical properties of MSW ash:

o 10% Portland Cement added to raw MSW ash will achieve liner-like permeabilities ranging from 7.5 X E-7 cm/sec to 2.8 X E-9 cm/sec, depending on permeability test method.

o Permeabilities of 6.6% Portland Cement and raw ash varied from 4.15 X E-7 cm/sec to 2.0 X E-8 cm/sec, depending on test method.

o 6.6% Lime [CaO] and MSW ash achieved permeability of 2.66 - 6.4 X E-6 cm/sec reducing the untreated ashes permeability by an order of magnitude suggesting a lime-based pozzolanic reaction.

o Costs for in-place addition of 10% Portland Cement range from $ 3,900 to $ 50,000 per acre, depending on landfill operator capability.

o Applying in-situ chemical treatment, if required, and an engineering methodology to MSW ash disposal should realize significant cost savings compared to single and double, synthetic and clay liners (costing from $ 250,000 to $ 500,000 per acre).

o The combination of proprietary mechanical processing and compaction of MSW ash could achieve volumetric reductions of 48% for MSW ash monofill land disposal requirements.

o Taking into account the approximate 25 % weight reduction, achieved from waste-to-energy systems, proprietary mechanical processing and compaction of MSW ash reduces land usage by 89 % to 93 % as compared to MSW landfill.

Engineering management of MSW ash monofills could consist of the elimination of daily cover as a regulatory requirement and the use of in-situ chemical treatment, if required, to achieve field permeabilities between 10 X E-6 cm/sec to 10 X E-9 cm/sec and high quality leachate/runoff.

The empirical results presented represents a significant investment of funds and time; reflecting a single private sector firm's commitment to the waste-to-energy field. Such expenditures of resources, achieving the reported results, signifies attaining a preeminent leadership role in resolving the controversies of MSW ash disposal. Others, both private and public, within the waste-to-energy field should follow this leadership example by expending their resources to similarly contribute in allaying the misconceptions associated with MSW ash disposal.

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AUTHORS' RESTRICTION

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TABLE 1. MINERALOGICAL COMPOSITION COMPARISON MSW ASH TO CEMENT

Composition of Portland Cement

Component	Cement	Clinker	MSW Ash
SiO_2	18-24	21.7-23.8	24
Al_2O_3	4-8	5.0-5.3	6
Fe_2O_3	1.5-4.5	0.2-2.6	3
CaO	62-67	67.7-70.8	37

NOTE: All values reported in Per Cent (%) by weight

TABLE 2. COMPARISON SIZE DISTRIBUTION AND COMPACTED DENSITY

Test Patch Placement	% Moisture (a)	Density (pcf) (a) Wet	Density (pcf) (a) Dry
Pre-compaction	18.15	123.52	104.58
Post-compaction	19.00	134.82	113.31

Ash	Dmax	Dmean	Dmin	Compacted DD (pcf)
BA (I)	30 mm	7.2 mm	0.04 mm	92.1
CA (I)	20 mm	3.0 mm	0.04 mm	74.9
CA (II)	12 mm	4.2 mm	0.04 mm	113.31

NOTE: (a) In-Place Test Patch Measurements
DD = Dry Density; pcf = pounds per cubic foot
mm = millimeters

TABLE 3. PERMEABILITY COMPARISON FIELD AND LABORATORY PROGRAM

FIELD PROGRAM PERMEABILITY RESULTS

Dosage	In-Situ	Laboratory Testing (Test Patch Ash)
CA (II) + 0%	-	1.9 X E-5
TP#1 CA (II) + 10%PC	Max 2.8 X E-9	7.5 X E-7 to Max 1.13 X E-8
TP#2 CA (II) + 6.6%PC	Max 3.53 X E-7	4.15 X E-7 to 2.0 X E-8
TP#3 CA (II) + 6.6%CaO	Max 6.4 X E-6	2.66 X E-6 to 2.3 X E-8

LABORATORY PROGRAM PERMEABILITY RESULTS

Ash	Additive (%)	Permeability
BA (I)	0%	1.8 X E-5
BA (I)	6% PC	1.5 X E-7
BA (I)	9% PC	1.7 X E-8
CA (I)	0%	5.5 X E-6
CA (I)	3% CaO	8.1 X E-7
CA (I)	6% CaO	4.2 X E-6

NOTES: All cores were tested at 28 days curing; Test Patch mixes and Laboratory Program testing were conducted at 14 days curing. Results reported in cm/sec.

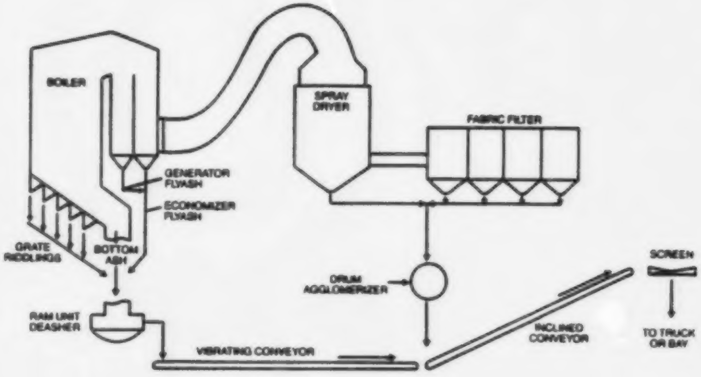
BOILER
SPRAY DRYER
FABRIC FILTER
GENERATOR FLYASH
ECONOMIZER FLYASH
GRATE RIDDLINGS
BOTTOM ASH
RAM UNIT DEASHER
DRUM AGGLOMERIZER
VIBRATING CONVEYOR
INCLINED CONVEYOR
SCREEN
TO TRUCK OR BAY

Figure 1. Ash Handling General Schematic

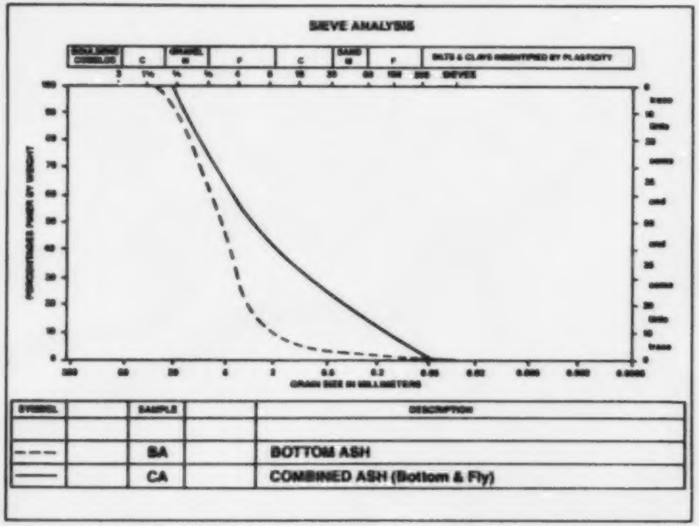

Figure 2. Particle Size Distribution Ash I (BA + CA)

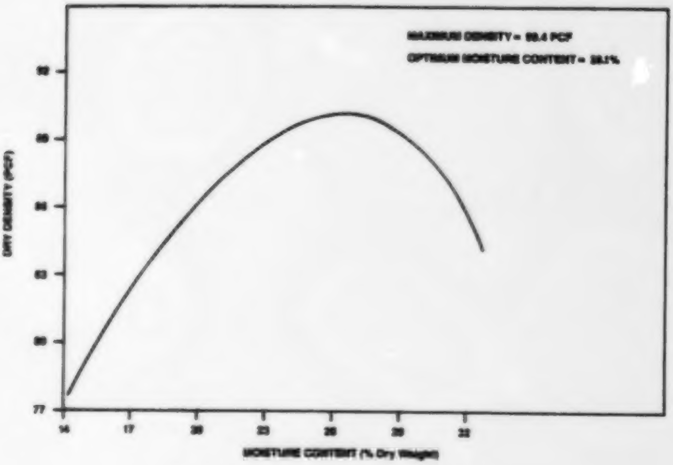

Figure 4. Compaction Curve Ash I (CA)

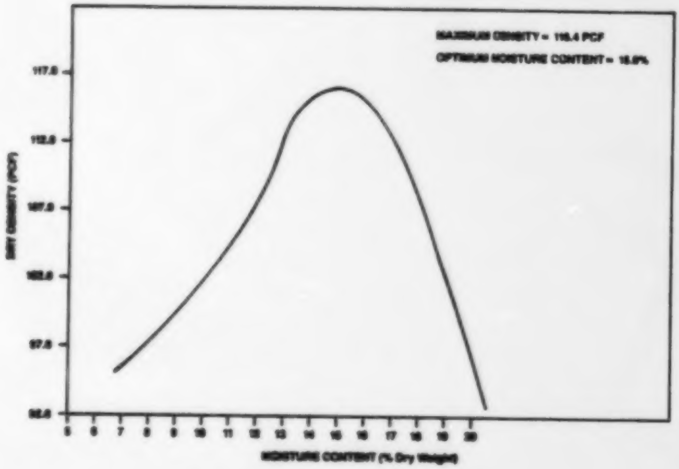

Figure 3. Compaction Curve Ash I (BA)

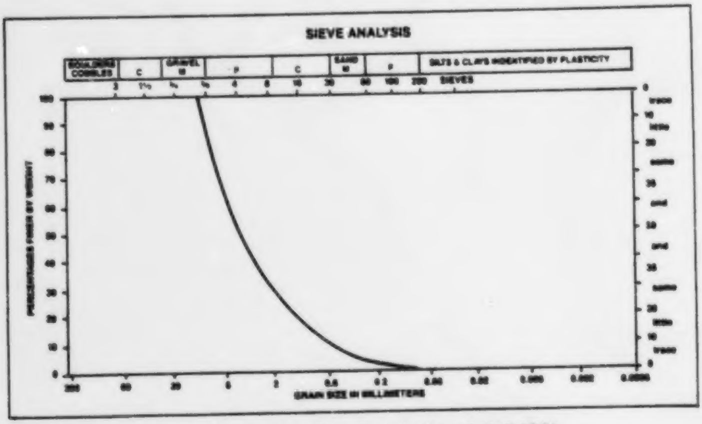

Figure 5. Particle Size Distribution Ash II (CA)

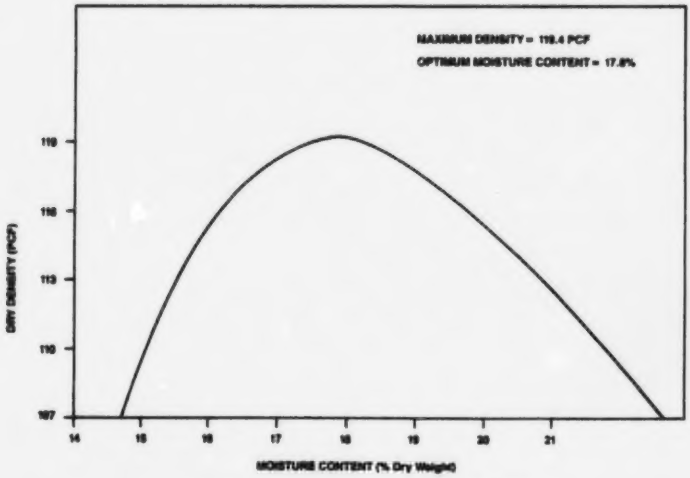

Figure 6. Compaction Curve Ash II (CA)

LOW TEMPERATURE TREATMENT OF FILTER ASH FROM MUNICIPAL SOLID WASTE INCINERATORS FOR REDUCTION OF DIOXINS AND RELATED COMPOUNDS

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ABSTRACT

In the temperature range of 250 to 400 °C filter ash from electrostatic precipitators of municipal waste incinerators shows in the presence of air ("oxygen surplus conditions") a high potential of dioxin formation by de novo synthesis. The starting reaction of this de novo synthesis of PCDD/PCDF is a chlorine formation from metal chlorides analogous to the Deacon-process. We could demonstrate that fly ash also has a high catalytic activity of PCDD/PCDF destruction via dechlorination/hydrogenation, which increases exponentially with temperature. In the temperature range of 250 to 400 °C the rate of PCDD/PCDF formation in the presence of oxygen is greater than the rate of PCDD/PCDF destruction by dechlorinaton. If, however, the PCDD/PCDF formation is limited by suppression of the chlorine formation due to a limited supply of oxygen ("oxygen deficiency conditions") the PCDD/PCDF concentration in filter ash can be reduced drastically by treatment in the temperature range of 300 to 400 °C.

The feasability of a technical application of this low temperature treatment of filter ash for reduction of dioxins and related compounds was tested on a technical scale in a pilot plant with a throughput of 60 kg/h. Reductions of PCDD/PCDF in the range of 95 to 99.9% were obtained. PCBs, chlorobenzenes and chloronaphthalenes were reduced to 93%.

At present a full scale plant is under construction at the municipal waste incinerator at Stuttgart for low temperature treatment of the filter ash from 40 t of waste per hour.

INTRODUCTION

About 10 years ago Olie et al. (1) reported on the detection of polychlorinated dibenzodioxins (PCDD) and polychlorinated dibenzofurans (PCDF) in electrostatic precipitator ash of municipal solid waste incinerators. Since then PCDD/PCDF ("dioxins") have been found worldwide in all ash samples from offgas cleaning devices of waste incinerators analyzed for dioxins. In the Federal

Republic of Germany the dioxin content of the electrostatic precipitator ash has led in the past to considerable problems with regard to their disposal.

In the last two years we have developed a procedure to reduce drastically the concentrations of dioxins and other chloroorganic compounds in filter ash by a thermal treatment in the temperature range of 350-400 °C.

BASIC LABORATORY EXPERIMENTS

The basis for the development of a procedure for dioxin reduction in filter ash were laboratory experiments in which we could demonstrate the catalytic effects of fly ash from waste incinerators in the formation and destruction of dioxins (2). We could show that the formation of dioxins at 250-400°C and their decomposition at 600 °C reported by Vogg and Stieglitz (3) is a superposition of two processes catalyzed by fly ash. For the formation of dioxins the catalytic formation of chlorine from copper and/or other metal chlorides in the ash particles in the presence of oxygen is the decisive starting reaction (2). The formation of dioxins by this mechanism is a *de novo* synthesis, a synthesis from "inorganic" chloride and "organic" carbon (4,5,6).

The important role of oxygen in the formation of dioxins became apparent when the thermal treatment of fly ash under "oxygen deficient" conditions in the temperature range of 250-400 °C did not result in dioxin formation. On the contrary, a drastic reduction of the dioxin concentrations was observed under these conditions. The mechanism of this dioxin decomposition is mainly a dechlorination/hydrogenation catalyzed by the fly ash (2,7).

The catalytic formation and decomposition of dioxins can thus be interpreted as a dualistic principle of *de novo* synthesis and dechlorination/hydrogenation (Figure 1) (8). Considering the chlorine formation as the rate limiting step in the *de novo* synthesis of dioxins one can assume that the rate of reaction as a function of temperature approaches saturation in the range of 250-400 °C. The rate of reaction for the catalytic dechlorination increases, how ever, exponentially with temperature.

This means that, depending on various factors (concentrations of chloride, organic carbon, heavy metals, sulfur, surface activity etc.), there can exist a temperature range in which on the ash particles by *de novo* synthesis more dioxins are produced than can be destroyed by dechlorination. The dechlorination reaction is an integral part of the dioxin formation in incinerators resulting in the characteristic isomer and homologue distribution found in filter ash and stack gas.

If the formation of chlorine and thus the formation of dioxins is prevented by limiting the supply of oxygen, the catalytic dechlorination leads to an almost complete destruction of PDD/PCDF in filter ash in the temperature range of 300-400°C (2). In Table 1 the decrease in dioxin concentrations of filter ash by thermal treatment at 300 °C under "oxygen deficiency" conditions in laboratory experiments (ampoules are sealed under atmospheric pressure) is demonstrated. That the

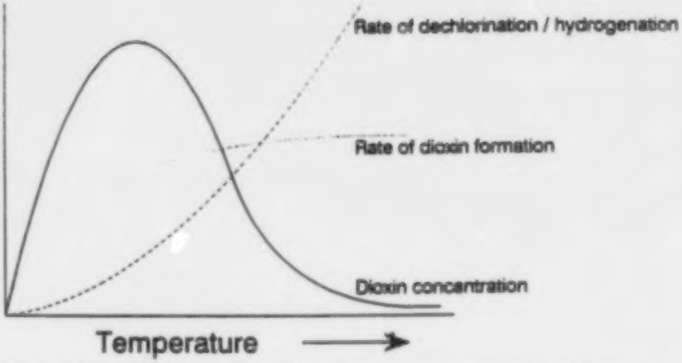

Figure 1. Schematic representation of the temperature dependence of dioxin formation and decomposition in the presence of oxygen

Table 1

Reduction of PCDD/PCDF-concentrations in electrostatic precipitator ash by thermal treatment under oxygen deficiency conditions in laboratory experiments.

Concentrations in ng/g (ppb). Detection limit for single isomers about 0.01 ng/g. n.d. = not detected.

A = untreated B = thermally treated at 300°C for 2h

	fly ash A		fly ash B		fly ash C	
	A	B	A	B	A	B
Tetrachlorodibenzodioxins	0.6	n.d.	7.5	0.2	5.2	1.1
Pentachlorodibenzodioxins	1.2	n.d.	13.2	0.3	11.4	n.d.
Hexachlorodibenzodioxins	1.4	n.d.	15.8	0.1	15.0	n.d.
Heptachlorodibenzodioxins	1.8	n.d.	12.4	0.1	25.6	n.d.
Octachlorodibenzodioxin	1.9	n.d.	14.5	n.d.	23.1	n.d.
Total PCDDs	**6.9**	—	**63.4**	**0.7**	**80.3**	**1.1**
Tetrachlorodibenzofurans	9.0	n.d.	28.7	0.2	22.4	10.1
Pentachlorodibenzofurans	10.2	n.d.	32.5	0.1	24.4	n.d.
Hexachlorodibenzofurans	8.0	n.d.	24.6	0.1	19.0	n.d.
Heptachlorodibenzofurans	3.4	n.d.	12.8	n.d.	11.1	n.d.
Octachlorodibenzofuran	0.7	n.d.	4.6	n.d.	2.1	n.d.
Total PCDFs	**31.3**	—	**103.2**	**0.4**	**79.0**	**10.1**

catalytic effect of low temperature dioxin decomposition is a general property of fly ash from waste incinerators has been shown for more than 100 samples of filter ash from various municipal and hospital waste incinerators.

FROM THE LABORATORY EXPERIMENT TO A TECHNICAL PLANT

In laboratory experiments we could demonstrate that the thermal treatment of filter ash at 300 °C under oxygen deficiency conditions leads to a decomposition of dioxins by a catalytic dechlorination reaction. The question arose whether the reduction of dioxin concentrations in filter ash by low temperature treatment under oxygen exclusion found in laboratory experiments could be realized also on a technical scale.

To test the possibility of a transfer to a technical scale a pilot plant was installed by the Technische Werke der Stadt Stuttgart (TWS) at the municipal waste incinerator in Stuttgart- Münster with the goal to verify and, if possible, to optimize the low temperature destruction of dioxins in filter ash.

A schematic drawing of this pilot plant is shown in Figure 2. The fly ash was heated under exclusion of air indirectly by the flue gas of the incinerator. The temperature could be varied from 250-450°C. The exclusion of air was either obtained by the filter ash itself or by valves (9).

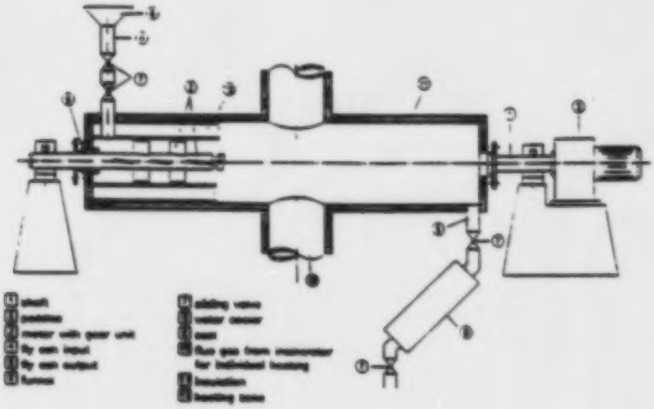
shaft
paddles
motor with gear unit
fly ash input
fly ash output
furnace
sliding valve
water cooler
seal
flue gas from incinerator for indirect heating
insulation
heating zone

Figure 2. Pilot plant for the thermal treatment of filter ash

The best decontamination results were obtained in the temperature range of 350-400 °C and residence time for the ash of 30 minutes to one hour. The reduction in PCDD/PCDF-concentrations ranged, depending on the starting concentrations, from 95% to 99.9%.

In Table 2 results from the thermal treatment of filter ash from the boiler 27 (low starting concentration) and boiler 29 (high starting concentration) of the Stuttgart incinerator in the pilot plant are shown.

Table 2
Efficiency of PCDD/PCDF-reduction by thermal treatment of fly ash in the pilot plant of the TWS AG in Stuttgart at 400°C for 1h.
Concentrations in ng/g. Detection limit for single isomers about 0.01 ng/g. n.d. = not detected
A = untreated B = thermally treated at 400°C for 1h

	fly ash boiler 27		fly ash boiler 29	
	A	B	A	B
Tetrachlorodibenzodioxins	5	0.2	42	n.d.
Pentachlorodibenzodioxins	13	0.1	94	n.d.
Hexachlorodibenzodioxins	19	n.d.	130	n.d.
Heptachlorodibenzodioxins	22	n.d.	76	n.d.
Octachlorodibenzodioxin	20	n.d.	73	0.3
Total PCDDs	**79**	**0.3**	**415**	**0.3**
Tetrachlorodibenzofurans	34	4	237	0.7
Pentachlorodibenzofurans	36	1	205	n.d.
Hexachlorodibenzofurans	20	0.1	87	n.d.
Heptachlorodibenzofurans	12	n.d.	44	n.d.
Octachlorodibenzofuran	3	n.d.	9	n.d.
Total PCDF	**105**	**5.1**	**582**	**0.7**

In a continous performance test of the pilot plant over 14 hours these results were confirmed. A total of 25 samples of thermally treated filter ash were analyzed for dioxins and other chlorinated aromatic compounds. In Figure 3 the total PCDD/PCDF-concentrations for the treated and the corresponding untreated filter ash samples are shown. The overall decrease in dioxin concentration for all samples amounts to 97.1% for PCDD and 97% for PCDF.

The main results obtained from the pilot plant experiments can be summarized as follows:

— Independant from the starting concentration more than 97% of the dioxins are destroyed by the low temperature treatment.

— After the treatment a low residual dioxin concentration remains which possibly can be further reduced by proper plant design.

— The thermal treatment also reduces other chlorinated aromatic compounds like chlorobenzenes, chlorophenols, polychlorinated biphenyls, chlorinated naphthalenes etc. to the same degree as described for dioxins.

— During the thermal treatment of the ash no pressure built-up in the pilot plant was observed.

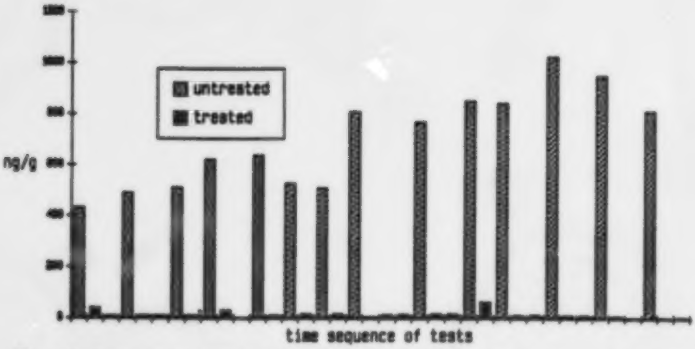

Figure 3. Comparison of total PCDD/PCDF concentrations of thermally treated and untreated electrostatic precipitator ash. Continuous test at the pilot plant of the TWS AG in Stuttgart. Sampling and analysis of corresponding untreated and thermally at 400 °C for 1 h treated ash of the same lot.

The low temperature decomposition of dioxins and related compounds shows the following characteristics:

- The thermal treatment of the ash is carried out in an indirectly heated system in the absence of a gas stream.
- The thermal treatment is carried out in the temperature range of 300- 400 °C. In this range no desorption of dioxins in the gas phase is observed in the absence of streaming gas.

- The process is carried out in a quasi-closed system, since the filter ash itself forms, after cooling to <100 °C, a seal towards the atmosphere.

- The efficiency of decomposition for individual compounds can be determined unequivocally by analysis.

From these characteristics the advantages of the described pro cedure compared to a treatment of the filter ash by reloading on the grate (11) become obvious. By a treatment of filter ash on the grate the ash goes in the presence of oxygen through the temperature zones of dioxin-formation (300-400 °C) and dioxin-desorption (in a stream of gas desorption starts at 350 °C) in an uncontrollable manner before it reaches a temperature were in the presence of oxygen an efficient destruction is observed. The fate of the dioxins in the filter ash and thus the real destruction efficiency is in this procedure not under analytical control.

The described investigations have shown that it is possible to reduce dioxins and related compounds, formed in the waste incinerator in the cooling zones by *de novo* synthesis (2,10) by technically simple thermal treatment of filter ash. These results were encouraging enough that now a full scale plant for the thermal treatment of electrostatic precipitator ash with a capacity of 600 kg/h to 1 t/h is being built by Deutsche Babcock at the Stuttgart waste incinerator, going into operation in June 1989.

FINAL CONSIDERATIONS

Solid residues from offgas cleaning of waste incinerators contain halogenated organic compounds which in their majority were not originally present as such in the waste itself. The better the stack emission of these compounds, formed in the incinerator, can be controlled by gas cleaning procedures the higher their concentration in the residues of the gas cleaning will be.

Solid residues from gas cleaning do not only contain these chloroorganic compounds formed in the incinerator but also heavy metals as further compounds of concern. There is however a distinct difference between the chlorinated compounds and the heavy metals. The heavy metals are already present in the waste and are by the incineration process only concentrated and transformed in their binding state. Other disposal options of municipal waste which result in a reduction of mass like landfill or composting also result in an increase in heavy metal concentration. The chlorinated organic compounds of concern are, however, in their majority produced in the incinerator. Even if one can assume that a mobilisation of these compounds from filter ash through the aquatic or atmospheric path is so low that it is analytically not detectable, "zero- concentrations" do not exist in this respect. Landfills with residues from gas cleaning which contain dioxins and other halogenated compounds represent indeed reservoirs for the global distribution of these compounds. The more new waste incinerators will go in operation the more important will become this reservoir for the global distribution of these compounds.

In the Federal Republic of Germany there have been discussions on limiting values for dioxin concentrations in filter ash and stack gases. Since there is no scientific base for such limiting values neither for solid residues nor for stack gas, as long as the toxicological assessment of dioxins has been completed, in Germany one has agreed that the principle of minimization on the basis of the technical standard and the economical feasability should be followed. Our investigations have shown that it is possible to reduce drastically dioxins and related compounds produced in the waste incinerator by a low temperature treatment. This procedure offers a real solution of the problem without any danger of a mere shift of the problem. The process of the low temperature decomposition of PCDD/PCDF and related compounds in filter ash contributes to the reduction of emissions of these potentially dangerous compounds. All these compounds are unwanted side products in the solid residues of waste incineration and should be decomposed in the incinerator if technically possible and economically feasible before they leave the incinerator, thereby also increasing the acceptance of urgently needed new municipal waste incinerators.

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LONG-TERM ASSESSMENT OF BOTTOM ASH MONOFILL LEACHATES

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ABSTRACT

Chemical reaction mechanisms of bottom ashes from municipal solid waste incinerators with water were investigated. Bottom ash composition, trace element concentrations, kinetics of dissolution reactions, alkalinity of bottom ash and field measurements were used to assess the long-term behavior of leachates of bottom ash monofills.

The results show that the organic carbon may be the most important proton producer from biological degradation of organic material to CO_2. The heavy metal leaching of bottom ashes in a monofill can be increased as a result of acidification of the monofill.

Laboratory experiments and field assessments support the hypothesis that bottom ash cannot be considered as a material of final storage quality, i.e., a physical chemical state of the monofill with emissions compatible with the environment. High non-metal concentrations in leachates can be expected over the next few decades after disposal.

Consequently, bottom ashes have to be pretreated to achieve the final storage quality before disposal. Otherwise the leachate has to be treated at least during the first few decades after deposition.

INTRODUCTION

In Switzerland, municipal solid waste incinerators produce nearly 600 000 tons (100 kg/c,c=capita) bottom ash per year (1,2). This mass flux corresponds to about 20-30% of the total annual municipal solid waste generated (3). Although the present disposal practice varies from case to case, a great deal of efforts is directed towards landfilling the bottom ash in monofills without mixing with fly ash or with other wastes.

One essential objective of the "guidelines to the waste management in Switzerland (4)" is that every waste treatment procedure has to produce materials which are either recyclable or can be landfilled without an adverse environmental impact for long-term periods. This kind of landfilling is defined as a "final storage". Materials disposed in a final storage have "final storage quality". Emissions from a final storage are compatible with the environment without any further treatment. Therefore, Swiss waste management policy has the objective to produce materials of final storage quality from non-recyclable solid wastes. These materials should have properties similar to the natural materials known in the earth's crust. They are

- natural sediments,
- stones and ores,
- soils.

Since the stabilities of these natural materials depend on the environmental conditions around the storage site, the fluctuations in physical, chemical and geological conditions in final storage must also be taken into account in long-term periods. The technical and scientific criteria of the final storage quality was discussed in detail in an international workshop, "The Landfill, Reactor and Final Storage", held in Switzerland in March 14-17, 1988 (5).

In a municipal solid waste incinerator, solid waste components undergo many physical and chemical processes and are distributed to flue gas, to flue gas cleaning residues and to bottom ash. This partitioning is determined by the process conditions in the incinerator and in the gas cleaning facilities. Due to high concentrations of many heavy metals, gas cleaning residues should be transformed to ore-like materials of final storage quality. On the other hand, bottom ash can be considered as a stone-like material.

Disposed in a monofill, bottom ash will contact with rain water. Experiences in Swiss landfill sites have shown that it is unlikely to prevent solid waste from contacting with rain water by encapsuling the landfill site with liners for long-term periods. As a result of bottom ash contact with water, many microbiological and chemical reactions can occur. Organic compounds can be transformed to other organic or inorganic compounds, such as organic acids, methane, carbon dioxide, etc. . Inorganic compounds can undergo several redox reactions. Furthermore, many dissolution, adsorption, precipitation processes can take place. Unfortunately, these physical, chemical and microbiological processes are very complex and cannot be easily simulated.

In this paper, a method is presented to investigate the main reactions of bottom ash with water with respect to long-term remobilization processes. The results are then used to discuss the final storage quality of bottom ash monofills.

METHODS

CONCEPT

A bottom ash monofill is a heterogeneous fixed bed reactor. It can be regarded as a combination of several reaction zones. In these zones many microbiological and chemical reactions occur, as well as various physical processes, such as adsorption, desorption, dissolution, precipitation, diffusion etc. . The reactants which enter the reactor are bottom ash and water. The products are the leachate and the remaining residual solid (in negligible amounts also gas). Non-reacted inputs and products of the complex reactions are transported by water which percolates through the reactor. This water flow is also heterogeneous. Consequently, it is almost impossible to simulate a real bottom ash monofill, in order to predict the leachate composition for long-term periods. Instead, basic investigations can provide very useful criteria to assess the long-term behavior of bottom ash monofills. These basic investigations include analysis of bottom ash , leaching experiments and a prediction of essential long-term processes in a monofill.

Bottom ash analysis involves determination of main compounds and trace elements, as well as the alkalinity. Leaching experiments are designed to provide profound understanding of reaction types and orders of magnitude of reaction rates. The results lead to the estimation of the "mobile" parts and "practically inert" parts of bottom ashes. With the assumption that the mobile parts of bottom ashes in the monofill are of the same order of magnitude as the values obtained in the laboratory, the long-term behavior of bottom ash monofills can be roughly predicted.

SAMPLING

Bottom ash which had passed a slag bath with a residence time of about 30-40 minutes was used for the experiments. It was mixed thoroughly and screened with a sieve to separate the sizes greater than 5 cm. The grains passed through the screen were then brought to the laboratory. In the laboratory, the bottom ash was dried for 24 h at 105 °C, ground for 2h in a steel bowl mill and screened again with a sieve. The resulting fraction had a particle size of less than 0.5 mm. It was then analyzed and used for all experiments.

REACTION WITH WATER

30 g of pulverized sample was mixed in an Erlenmeyer flask with 300 ml distilled water and the flask was covered with parafilm. This two-phase system was shaken on a rotary type extractor at 150 rpm for 0.5, 5, 50 or 500 hours. The suspension was filtered through a 0.45 µm membrane filter at the end of the extraction time. Concentrations of main cations and anions were determined in the extracts, as well as total concentrations of some elements and some bulk parameters (e.g., dissolved organic carbon (DOC)). An additional extraction was performed analogously with the filtration residue.

TITRATION OF BOTTOM ASH

In order to assess the acid/base reactions in the monofill, the alkalinity of the bottom ash has to be known. It is defined operationally by the amount of the acid required to reach pH 7. Eight parallel experiments were carried out with 2 g pulverized bottom ash in 100 ml aqueous solution with increasing aliquots of HNO_3 to determine the alkalinity. The pH of the suspension was measured after 48 hours of rotary-type shaking (150 rpm) in an Erlenmeyer flask.

RESULTS AND DISCUSSION

COMPOSITION OF BOTTOM ASH

Table 1 shows the composition of bottom ash assuming all main elements (>1% w/w) are in oxidized form. The most abundant main element is silicon. More than 50% of the bottom ash consists of oxides of silicon. The second group of elements which exist in high concentrations are calcium, iron and aluminium. Sodium, magnesium and potassium are the other main elements. Consequently, more than 95% of the matrix consist of the main elements listed in this Table.

TABLE 1. Composition of bottom ash, granite and basalt

	Bottom Ash	Granite		Basalt	
	Conc. [%]	Conc. [%]	Enrichment factor[a] [-]	Conc. [%]	Enrichment factor[a] [-]
SiO_2	51	71	0.7	50	1.0
CaO	15	2	7.5	10	1.5
Fe_2O_3+FeO	12	3	4.0	12	1.0
Al_2O_3	11	14	0.8	15	0.7
Na_2O	3	3	1.0	3	1.0
MgO	2	1	2.0	7	0.3
K_2O	2	5	0.4	1	2.0
Total	96	99		98	

[a] $$\text{Enrichment factor} = \frac{\text{Concentration in Bottom Ash}}{\text{Concentration in Rock}}$$

The composition of bottom ash can be compared with igneous rocks, e.g., granite and basalt. Granite is a coarse grained, slowly cooled igneous rock which is relatively rich in silicon. Basalt is a fine grained, quickly cooled igneous rock which contains abundant iron and magnesium. Typical average compositions (more than 100 samples) of these rocks are given in Table 1 (6,7).

The silicon and iron contents of bottom ash are similar to basalt. Granite contains more silicon and less iron than bottom ash. Calcium content of bottom ash is higher and aluminium content lower than the calcium and aluminium contents of both rocks. The abundance of sodium, magnesium and potassium in bottom ash are between granite and basalt. However, these main elements exist in bottom ash in the same order of magnitude (10> Enrichment factor >0.1) as in the igneous rocks. Consequently, bottom ash can be defined as a stone-like material.

Table 2 shows some trace element concentrations in bottom ash and average concentrations in igneous rocks of the upper continental crust (7).

TABLE 2. Trace element concentrations in bottom ash and average concentrations in igneous rocks of the upper earth's crust

	Concentration in Bottom Ash [mg/kg]	Average Conc. in Igneous Rocks [mg/kg]	Enrichment factor [-]
C	16000	300	50
S	5000	300	15
P	5000	800	6
Cl	3000	300	10
F	300	700	0.4
Cu	1000	30	30
Zn	3000	60	50
Pb	1300	15	90
Cd	8	0.1	80
Hg	0.3	0.03	10

The non-metals are enriched in bottom ashes about 5-15 times compared with igneous rocks (fluorine is an exception). Metal enrichment is even higher (enrichment factor = 10-90). Consequently, even tough bottom ash can be considered as a stone-like material, trace elements may become important with regard to the final storage quality of bottom ash. Special attention should be given to them in order to assess the long-term behavior of bottom ash in monofills.

REACTION WITH WATER

Figure 1 illustrates the concentrations of the main cations calcium, sodium and potassium as a function of time in the leaching experiments. The concentrations in the extracts increase with increasing reaction time. From 50 to 500 hours, however, the concentrations of these cations are either constant or slightly increasing.

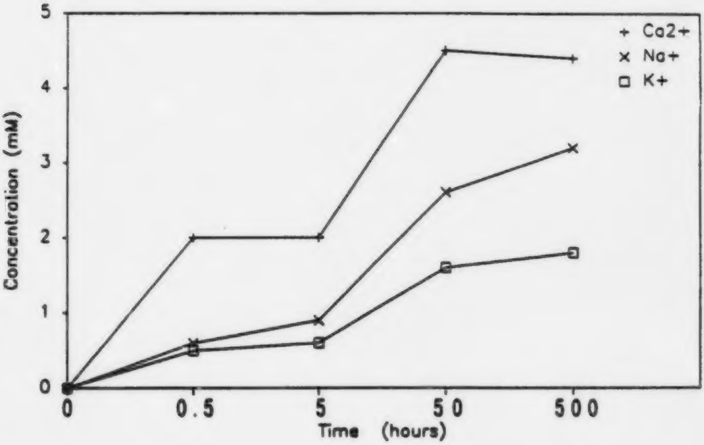

Figure 1. Kinetics of the dissolution reaction of main cations from bottom ash

Figure 2 gives the kinetics of the leaching experiments of the main anions. In general, the concentrations increase with increasing time. This is valid for a period of up to 50 hours. Concentrations of chloride and sulfate keep increasing during between 50 h and 500 h as well. Hydroxide concentrations decrease during the same time period. This might be due to formation of the $Al(OH)_4^-$ from $Al(OH)_3$ (8).Concentration calculations carried out with the complex formation constants confirm this behavior.

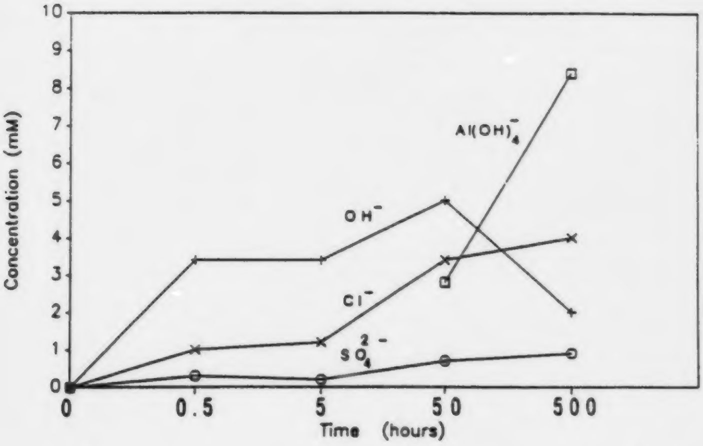

Figure 2. Kinetics of the dissolution reaction of main anions from bottom ash

Figure 3 shows ion balances in the extracts. In the first extraction, the total concentrations of the main cations amount to about 13 miliequivalent per liter. The total concentrations of the main anions are nearly equal to the total concentrations of the main cations. The total concentrations in the additional extraction, which is carried out with the residual of the first extraction, are slightly lower. The calcium ion is again the main cation. But hydroxide is replaced by sulfate ion. The dissolution of sulfates is probably slower than the formation of main hydroxides. The pH values in the first extraction amount to about 11.5 and in the second extraction about 9.5. Another important observation is that the solution is not saturated with any species. The leachate composition is determined by the leaching kinetics, rather than by the saturation of the leachate.

The results of these experiments point out that a single leaching test conducted over a few hours or a few days may lead to some wrong results.

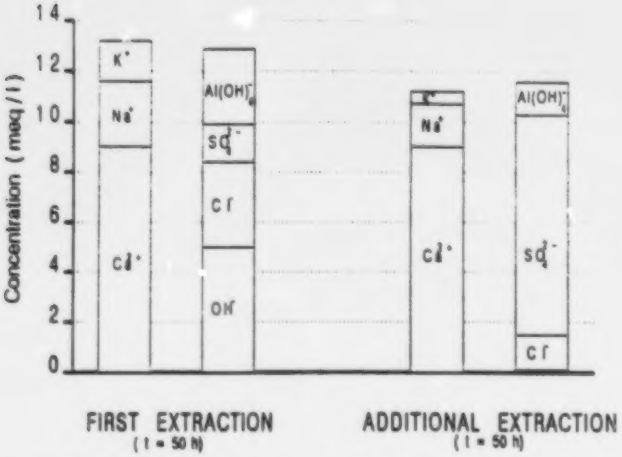

Figure 3. Ion balances in the extracts

ASSESSMENT OF THE LONG-TERM BEHAVIOR

After determining the properties of bottom ash and elucidating the reactions, the results,as a third step, should be combined to predict the long-term behavior of bottom ash monofills. The results of this prediction must be then interpreted in orders of magnitude.

One of the most important parameters determining the long-term remobilization processes is pH. A decrease of pH to less than 7 may mobilize many metals in monofills. Total organic carbon concentration (TOC) in the bottom ash is about 10 g C/kg bottom ash. Provided that still 50% of organic carbon can be oxidized to CO_2, less than 1 mol of weak acid can be produced per kg bottom ash (9). In a monofill with 10 m depth, a precipitation of 1000 mm per year with a pH of 4 would provide a proton flux of about 10^{-5} mol per kg bottom ash per year . In a time period of 1000 years, this proton source would be still two orders of magnitude lower than the proton potential of the bottom ash itself.

The alkalinity of the bottom ash, determined in eight parallel experiments, is about 1.5 mol/kg bottom ash. It is not much higher than the proton potential of the bottom ash. Consequently, in a long-term consideration, at least parts of the monofill could be acidified by CO_2-production. However, remobilization processes should occur in alkaline conditions during at least the next decades after disposal.

The measured total amounts m_M of 3 non-metals and 3 metals mobilized in two consecutive laboratory experiments and a comparison with the total concentrations m_B in the bottom ash are given in Table 3. Different elements show different remobilization characteristics. In general, the mobile parts of the non-metals are higher than the mobile parts of the metals. Assuming that the mobile parts of the elements in bottom ashes deposited in monofills are of the same order of magnitude as in the laboratory experiments, these values can be used for the prediction of the long-term behavior of the concentrations in leachates of bottom ash monofills.

TABLE 3. Total element concentrations m_B in bottom ash and element concentrations m_M in bottom ash that can be mobilized

	m_B	m_M	
	[mg/kg]	[mg/kg]	[%]
TOC[a]	10000	1000	10
Cl	3000	1600	55
S	5000	1500	30
Cu	1000	15	1.5
Zn	3000	6	0.2
Pb	1300	2	0.2

[a] TOC: total organic carbon

For a zero order remobilization rate, the expected element concentrations in the leachate are given by the following equation:

$$c_C = \frac{m_M}{Q_L * t}$$

where c_C is the annual mean element concentration in the leachate in mg per liter, m_M the element concentration in the bottom ash that can be mobilized in mg per kg bottom ash, Q_L the annual mean leachate flow in liter per kg bottom ash per year and t the leaching time in years.

A monofill with 10 m depth, a precipitation of 1000 mm per year and a 50% yield in leachate generation would produce approximately 0.05 liter leachate per kg bottom ash per year. ($Q_L = 0.05$ liter kg^{-1} $year^{-1}$). According to their mobilities (10), it is estimated that non-metals will be leached between 10 and 100 years and metals between 100 and 1000 years. The concentrations can be then calculated for these hypothetical leaching times.

The calculated concentrations c_C are in the same order of magnitude as in the concentration ranges c_L measured in the bottom ash landfills (Table 4) (11). Therefore, the estimated ranges for leaching times seem to be plausible.

TABLE 4. Assessment of the element concentrations in leachates of bottom ash monofills

	t [years]	c_C [mg/l]	c_L [mg/l]	c_{QS} [mg/l]
DOC	10-100	2000-200	50-500	20
Cl	10-100	3200-320	1000-10000	100
S	10-100	3000-300	100-1000	30[a]
Cu	100-1000	3-0.3	<0.01-0.1	0.1
Zn	100-1000	1-0.1	<0.05-1	0.6
Pb	100-1000	0.4-0.04	<0.02-0.5	0.5

The last column in Table 4 represents the quality standards c_{QS} chosen on the basis of the Swiss quality standards (10,12) for running waters (≤10 times of standard values). The non-metal concentrations expected in the next decades after disposal are higher than the chosen quality standards. The metal concentrations are in the same order of magnitude as the quality standards.

These results indicate that the mobile part of chlorine would be washed out after a few years, since the chlorine concentrations measured in leachates c_L of bottom ash landfills are in the upper range of the calculated concentrations c_C. Higher DOC and sulfur concentrations than the quality standards are expected over the next few decades. The leaching of the mobile parts of metals could last over centuries. However, this assessment is valid for pHs higher than 7.

CONCLUSIONS

Many physical, chemical and microbiological reactions can occur as a result of bottom ash contact with water in bottom ash monofills. Laboratory experiments show that the important reactions between bottom ash and water last at least days or even weeks. Therefore, one leaching test conducted in hours or days is insufficient to assess the bottom ash behavior in monofills.

Acid/base reactions in the monofill may play a highly important role for the remobilization of heavy metals. Therefore, the degree of the alkalinity of bottom ash has to be compared with the proton production potential of bottom ash. The organic carbon may be the most important proton producer due to degradation to CO_2. TOC-contents of higher than 1% could suffice to acidify at least parts of heavy metals. This hypothesis is supported by laboratory experiments, as well as field observations (9).

A first approach to assess the long-term behavior of bottom ashes in monofill consists of three steps:

1. Determination of the composition of the bottom ash, trace element concentrations and alkalinity;
2. investigation of the bottom ash reactions with water; and,
3. prediction of possible long-term processes and assessment of the leachate composition as a function of time.

The results show that higher non-metal concentrations are expected than the quality standards in leachates of bottom ash monofills over decades after disposal. If a quality standard chosen on the basis of Swiss quality standards for running waters has to be reached, leachates of bottom ash monofills have to be treated at least during several decades. Consequently, bottom ash cannot be considered as a material of final storage quality. However, it can be pretreated before disposal to achieve this quality.

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ACKNOWLEDGEMENT

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MEASUREMENT METHODOLOGY FOR TOXIC METALS FROM MUNICIPAL WASTE COMBUSTORS

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ABSTRACT

The U. S. Environmental Protection Agency is concerned with toxic metal emissions from stationary combustion sources. In the event toxic metal emissions are regulated, a measurement methodology will be needed. A methodology for measurement of multiple trace metal emissions has been developed and evaluated. Based on a literature search, a modified EPA Method 5 train using a quartz-fiber filter followed by a mixture of nitric acid and hydrogen peroxide in each of the first two impingers and acidified potassium permanganate (for mercury collection) in the following one or two impingers was selected for sample collection. Inductively coupled argon plasma emission spectroscopy (ICAP) and atomic absorption spectroscopy (AAS) were selected for analysis. Parr Bomb® and microwave sample digestion procedures were developed using standard reference materials and spiked samples with analysis by ICAP, AAS, and neutron activation. The candidate methodology was evaluated and developed further on the basis of two field evaluation tests conducted at a municipal waste incinerator and a sewage sludge incinerator. Quadruplicate-train sampling was used to produce multiple isokinetic samples for assessing method precision. Both the front half (filter and probe) and back half (impingers) samples were analyzed separately by ICAP and AAS. The field test results were used to evaluate the proposed methodology with respect to collection efficiency and precision for the 12 metals detected. Because of the mercury findings in the second test, a second acidified permanganate impinger has been added to the recommended sampling train.

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INTRODUCTION

The United States Environmental Protection Agency (EPA) is presently considering the health effects of metal emissions from municipal waste incinerators, sewage sludge incinerators, and hazardous waste incinerators. Should such emissions be regulated, appropriate methods must be available for their measurement.

A study was sponsored by the Atmospheric Research and Exposure Assessment Laboratory of the EPA to develop and evaluate a sampling and analytical protocol for measurement of multiple metals in emissions from stationary sources. The metals targeted for measurement were: antimony (Sb), arsenic (As), barium (Ba), beryllium (Be), cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), manganese (Mn), mercury (Hg), nickel (Ni), phosphorus (P), selenium (Se), silver (Ag), thallium (Tl), and zinc (Zn).

The objectives of the study were to (1) develop a sampling and analytical protocol for measuring stationary source emissions of multiple metals; (2) evaluate the collection media and sample preparation procedures in the laboratory; (3) evaluate absorbing reagent strength and sampling train collection efficiency in the field; (4) determine distribution in the sampling train for metals collected under field conditions; (5) determine the precision of the combined sampling and analysis protocol; (6) compare the collection of mercury by the sampling train to that of the Method 101A train; and (7) establish the method's detection limits for measuring the target metals expected in incineration process flue gas emission samples of typical Method 5 stack gas volumes (1.25 m^3).

A literature search was the basis for the sampling and analytical protocol that was developed. A modification of the Method 5 sampling train was selected for its ease of operation, availability of equipment, acceptable particulate collection efficiency, variable flow rate operation, easy modification to prevent contamination, adaptability to high-temperature sampling, and reasonable cost. A laboratory study was conducted to refine the sample digestion procedures, evaluate the collection media, and determine the accuracy for sample preparation and analysis. Finally, two field evaluation tests were conducted, one at a municipal waste combustor and the other at a sewage sludge incinerator, to evaluate and refine the sampling and analysis protocol with respect to impinger reagent strength, collection efficiency, method detection limits, and the precision of combined sampling and analysis. Additionally, the relative accuracy for collection of mercury as compared to Method 101A was assessed.

SELECTION OF CANDIDATE METHODOLOGY

The sampling and analytical protocol proposed and evaluated was based on a thorough literature review. Metals from incineration processes may be emitted in both particulate and volatile forms. Metals that have exhibited volatile behavior under past sampling conditions are mercury and arsenic.[1,2] Thus, development of a sampling system capable of collecting metals in both forms was necessary.

The Source Assessment Sampling System (SASS) and a modified Method 5 train were the primary candidates for sample collection.[3] Further evaluation indicated that the modified Method 5 train would be more suitable. Unlike the SASS train, the modified Method 5 train is easily adapted to include all-glass sample exposed surfaces to prevent contamination from stainless steel, etc.[3,4,5] In addition, it is capable of sampling at variable flow rates and temperatures to 260°C without modification. Finally, the modified Method 5 approach can produce a sample of 1 to 4 m^3 in a 1- to 2-h period which should be adequate for trace metals measurement.

A high-purity quartz-fiber filter (Pallflex® Type 2500 QAT-UP UltraPure) was selected for its low metal background values. For collection of metals reaching the impingers, a solution of 5% nitric acid (HNO_3)/10% hydrogen peroxide (H_2O_2) was selected to capture the majority of these emissions. Nitric acid has commonly been used to collect metals in solution.[6] Hydrogen peroxide was chosen to remove sulfur dioxide to prevent loss of oxidative capability in the first two collecting impingers and succeeding impingers, and it has been shown to capture 20% of the total mercury in a previous test.[7] Originally, a 1.5% potassium permanganate ($KMnO_4$)/10% sulfuric acid (H_2SO_4) solution was selected for use in the final absorbing impinger to capture the remainder of the mercury emissions, on the basis of good collection efficiencies shown previously by acidified permanganate solutions.[7,8] The train initially proposed also included a water impinger and an empty impinger following the HNO_3/H_2O_2 impingers. It was concluded, however, that these would unnecessarily complicate the train, because the HNO_3/H_2O_2 would most likely offer adequate capture of As, and careful use of the train would prevent transfer of the $KMnO_4$ solution.[9]

The analytical methods selected were atomic absorption spectroscopy (AAS) and inductively coupled argon plasma emission spectroscopy (ICAP). Other analytical methods were considered but were not selected because of the following considerations: ICAP is an acceptable method for the analysis of most elements because of its low cost, acceptable sensitivity, and multi-element analysis capabilities. ICAP can be combined with AAS for analyzing elements such as Hg, As, Se, and Pb, for which ICAP is less sensitive. General instrument availability is a factor in choosing an analytical method, and instruments for ICAP and AAS are generally more available than those for X-ray fluorescence or neutron activation analysis (NAA).

Based on the minimum detection limits for ICAP and AAS, and assuming a typical sampling time of 1 h and volume of 1.25 m^3, this method combination and the following analytical scheme could be used to detect the metals as described in the last section of this paper. Samples are first analyzed by ICAP for all elements except mercury. Cold vapor AAS is used for the analysis for mercury. If some metals are not found by the ICAP analysis or are found at levels at or near the detection limits, and if lower levels of detection are desired, the samples may be reanalyzed for these elements by using AAS. If such reanalysis is necessary, selenium and arsenic should be analyzed by either the graphite furnace or hydride method. Certain emissions tests may require further analytical design to yield lower method detection limits. For example, NAA could be used to supplement the ICAP/AAS method if NAA is available and proper standards can be obtained.

Digestion techniques were selected and designed to be compatible with the recommended analytical approaches. An acid digestion (nitric or hydrofluoric acid) using conventional or microwave heating techniques was proposed.[10] In particular, hydrofluoric acid (HF) was identified for dissolving any silica residues present.[11]

EXPERIMENTAL PROCEDURES

In the laboratory evaluation, the HNO_3/H_2O_2 absorbing solutions were spiked with the target metals at levels of 0.5 to 25 ppm. Samples were evaporated to dryness and digested by using conventional and microwave open vessel heating techniques. In both digestions 50% HNO_3 and 3% H_2O_2 were added with final dilution to 100 mL prior to analysis. The high-purity quartz-fiber filters were similarly spiked and digested with conventional (Parr Bomb®) and microwave pressure relief vessels with the same digestion media (4 mL each of concentrated HNO_3 and HF). Additionally, filters with 0.5 g fly ash added were digested by using the microwave digestion. Conventional digestions took 1 to 6 h and the microwave digestions took 0.1 to 0.3 h (6 to 18 min). All samples were analyzed by using AAS, ICAP, or NAA. Graphite furnace AAS was used for analysis of low levels of arsenic, selenium, and lead; ICAP was used for all other metals except mercury. In the analysis of other metals, interference caused by high iron and aluminum in some samples required dilution of the samples to 50 ppm for these two elements to eliminate their interference. The cold vapor AAS (CVAAS) technique was used to analyze for mercury.

The retention of metals in the absorbing solutions was evaluated by spiking a prototype impinger train, pulling ambient air through it at approximately 10 L/min for 2 h (yielding a total of 1.3 m^3, typical of Method 5 sampling volumes), acid digesting the samples by using microwave techniques, and analyzing by ICAP and CVAAS. Finally, the ability of the permanganate absorbing solution to hold collected mercury was evaluated; solution samples were spiked and later analyzed by CVAAS at various time intervals.

The first evaluation field test utilized single-point, collocated, quadruplicate-train (quad-train, see Figure 1) sampling to determine (1) if absorbing solution strength would affect collection of the target metals, (2) the relative accuracy of the proposed measurement method versus Method 101A for mercury, and (3) the distribution in the sampling train for the target metals. The quad-train sampling was performed at a municipal solid waste incinerator with electrostatic precipitator (ESP) emission controls and a stack gas temperature of 210°C. Two quad-train runs involved two multiple metals trains with regular and two with reduced strength nitric acid/hydrogen peroxide absorbing solution; two other quad-train runs involved two multiple metals trains and two Method 101A trains. All metals trains were used to assess sample distribution within the train. The multiple metals train consisted of an all-glass nozzle and probe assembly and a quartz-fiber filter followed by a series of five impingers and a Method 5 meter box and vacuum pump. The impinger sequence was as follows: an empty condensate impinger, two impingers with 5% HNO_3/10% H_2O_2, one impinger containing 1.5% $KMnO_4$/10% H_2SO_4, and an impinger containing silica gel. Sampling runs were conducted by using Method 5 procedures, were 4 h in length, at approximately 17 L/min, and produced sample volumes of approximately 4 dry standard cubic meters (dscm).

The second evaluation field test, which was conducted at a sewage sludge incinerator, also utilized single-point quad-train sampling; the multiple metals trains used were identical to those used for the first test, except that the condensate impinger was eliminated and the strength of the $KMnO_4$ was increased to 4% following the first run. Four quad-train runs with two standard metals trains and two metals trains with an extra HNO_3/H_2O_2 impinger were used to evaluate collection efficiency, sample distribution, and precision of sampling and analysis for each of the metals. Two other quad-train runs involving two metals trains and two Method 101A trains were used to determine the relative accuracy of the proposed train for collection of mercury. Sampling was performed isokinetically, with run times of 2 h and sample volumes of approximately 3 dscm.

Sample recovery was similar for the two field tests. The probe was rinsed with acetone followed by 0.1 N HNO_3. The HNO_3/H_2O_2 impingers (and condensate impinger for the first test) were recovered and rinsed with 0.1 N HNO_3. The $KMnO_4$/H_2SO_4 impinger was recovered and rinsed with fresh $KMnO_4$/H_2SO_4 of the same concentration. The filter was placed in a glass container.

All filter samples were acid digested by using closed-vessel microwave or closed-vessel conventional heating techniques. The front half (nozzle, probe, filter) rinse samples were acid digested by using closed-vessel microwave or conventional techniques for the first test and open-vessel microwave techniques for the second test. The HNO_3/H_2O_2 samples were acid digested by using open-vessel heating. Aliquots which were taken from the front half samples and from the HNO_3/H_2O_2 samples for mercury analysis plus the $KMnO_4$/H_2SO_4 samples were digested with acid and permanganate at 95°C for 2 h. The $KMnO_4$ samples and other aliquots taken for Hg analysis were analyzed for Hg by using Method 7470 in SW-846.[10] ICAP and AAS, as applicable on the basis of detection limits, were used to analyze separately the front half rinse samples, the filter samples, and the HNO_3/H_2O_2 samples for the remaining target metals.

RESULTS AND DISCUSSION

The laboratory studies showed that for the conventional heating digestion of the impinger contents, recoveries of the metals ranged from 90.8 to 104.6% with a mean overall recovery of 94.8%. Recoveries for the microwave heating digestion ranged from 88.2 to 109.1% with a mean overall recovery of 98.7%. An analysis of variance comparing the two digestion procedures showed a significant effect due both to the element analyzed and the digestion method used, but the interaction term between these two effects was not significant. Thus, although the percent recovery was different from element to element and between methods, the trends were in the same direction and were acceptable in percent recovered as shown.

For the conventional heating digestion of the spiked filters, recoveries of the spiked metals ranged from 84 to 107%. The average mean recovery was 95.2%. For the microwave heating digestion of these filters, recoveries of the spiked metals ranged from 85.7 to 101.1%, with an average recovery of 94.6%. A statistical comparison of the two methods showed no significant effect due to the digestion method used. These recoveries were acceptable as shown.

The analysis of the spiked fly ash samples in general showed lower mean recoveries than the spiked filters containing no fly ash. However, average recoveries of greater than 75% were obtained with the spiked fly ash samples for all elements except beryllium. The mean beryllium recovery was 54%. A possible explanation for the low beryllium recoveries is that beryllium may form insoluble complexes that precipitate out of the sample solution. Except for the beryllium, these recoveries were acceptable as shown.

Figure 2 shows the data for the microwave heating digestions, which are representative of the spiking experiments summarized above. The spiking experiments confirmed the efficacy of the digestion techniques proposed: for filter catches, hydrogen fluoride and nitric acid in a Parr Bomb® or pressure relief vessel heated in a conventional oven or a microwave, respectively; and for absorbing solution catches, nitric acid and hydrogen peroxide in an open vessel heated either conventionally or by microwave oven. When a precipitate is present and/or if undissolved particulate matter exists in the absorbing solution catches, these, like the filter catches, must be digested using hydrogen fluoride and nitric acid.

The retention experiment indicated that the absorbing solutions were able to retain most of the metals under simulated sampling conditions; all elements spiked showed greater than 90% recovery for all spiked elements except arsenic (83%).

The effect of HNO_3/H_2O_2 reagent strength was examined by using the results of two quad-trains from the first field evaluation. A two-way analysis of variance for Pb, Zn, As, Mn, and Hg showed no significant difference between the two solutions for metal collection.

The distribution of the metals in the front half (nozzle, probe, filter) and back half (filter frit, back half of the filter holder, impingers) of the trains was determined using the results for both field tests (see Table 1). Nine of the metals (Sb, Cd, Cr, Cu, Pb, Ni, Se, Tl, and Zn) were found almost exclusively (> 95%) in the front half. Hg was found almost exclusively in the back half. The distribution results for Ar, Ba, Be, Mn, and P varied between the two tests and Ag was not detected.

The analytical results for the extra (third) HNO_3/H_2O_2 impinger used in two multiple metals trains in four of the quad-train runs during the second test were used to determine the collection efficiency of the train. All metals, except Hg, Mn, and Ba were either not detectable in this impinger or present in amounts less than 1% of the total of that metal caught in the train, yielding a collection efficiency of approximately 100% for the entire train.

Results of the comparison of mercury collection by the multiple metals train and the Method 101A train are presented in Table 2. An analysis of variance conducted on the data from the first field test indicated that for measurement of mercury, no difference exists in the multiple metals train and the Method 101A train. However, for mercury, the results of the second field test indicated a low bias of about 30% for the multiple metals train versus the Method 101A train. In the first field test, about 90% of the mercury catch was found in the nitric acid/hydrogen peroxide impingers, whereas in the second field test 95% was found in the permanganate impingers. In several cases during the second field test, the color of the permanganate solution following testing indicated excessive decomposition of the permanganate, and therefore, the potential for inadequate mercury capture. Because of these results, the recommended train includes a second permanganate impinger to assure adequate mercury capture.

Finally, the results of the four quad-train runs involving four multiple metals trains conducted during the second field evaluation test were used to calculate the combined precisions for sampling and analysis. These are presented in Table 3 in terms of the relative standard deviation for the four trains in each run. When analyzed by ICAP and GFAAS in the range of five times the analytical detection limit, the average precision was between 11.2 and 15.3% for all metals except Ba, Be, Mn, Ni, and Ag, which were not detected or were below the quantifiable detection limit.

As part of the second field evaluation, a gravimetric determination was conducted on the particulate matter on the filter and in the acetone rinse. The particulate emission concentrations calculated were typical for controlled sewage sludge incinerators and ranged from 10.3 to 20.1 mg/dscm. The standard deviations (precisions) for the four trains in the four quad-train runs for which the particulate determinations were conducted were 15.7, 3.0, 6.2, and 7.3%. These are typical of the values associated with particulate sampling.

CONCLUSIONS AND RECOMMENDATIONS

The candidate protocol has yielded a precision (11.2 to 15.3% for the majority of target metals), collection efficiency (99+% for all target metals except Hg, Mn, and Ba), and in-stack detection limits for the measurement of the emissions of multiple metals using one train. The in-stack detection limits are discussed in detail below. On the basis of the mercury results from the second field test, the recommended sampling train includes a second permanganate impinger (see Figure 3), and the concentration of the permanganate has been increased to 4%, which is designed to eliminate the effect of oxidizable materials on the collection of mercury in the permanganate impingers. If testing experience can demonstrate that only one permanganate impinger is needed for a particular source type, then it will be acceptable. Additionally, the sampling procedures can be used to determine particulate emissions in the front half in conjunction with the metal emissions if the modified sample recovery procedures described in the draft method are followed.

Based on a 1.25-m^3 stack gas sample and liquid digest sample volumes of 300 mL for the front half and 150 mL for the back half of the train, the in-stack method detection limit for any specific metal is within the range from 0.03 to 27 ug/dscm for the train (0.02 to 18 ug/dscm for the front half and 0.01 to 9 ug/dscm for the back half). Table 4 shows the specific in-stack method detection limit for each metal.

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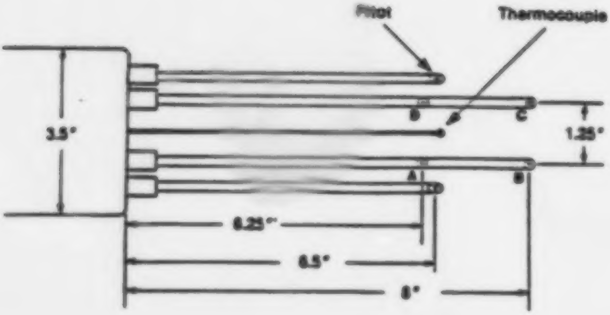
Pitot
Thermocouple
3.5"
1.25"
D
C
A
B
6.25"
6.5"
8"

Figure 1. Quad-train sampling probe design.

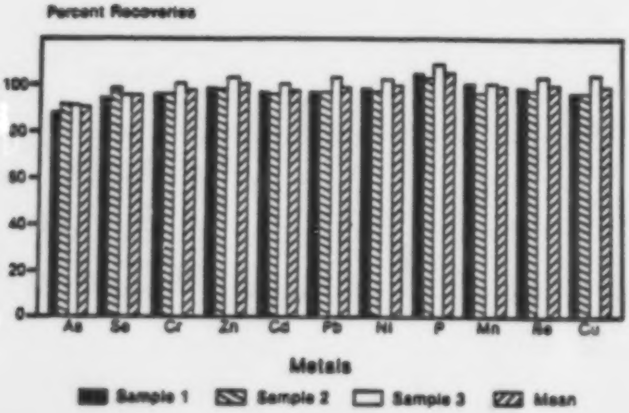

Figure 2. Percent recovery of spikes for microwave heating digestions (approx. 13 min).

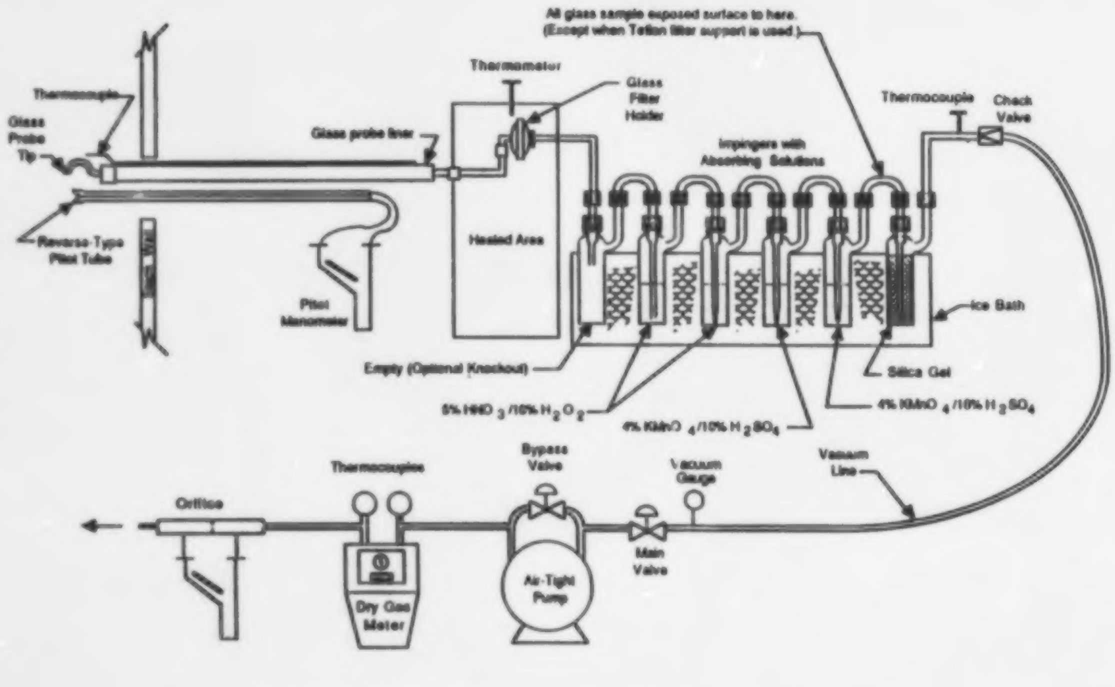
All glass sample exposed surface to here.
(Except when Teflon filter support is used.)
Thermocouple
Glass Probe Tip
Glass probe liner
Thermometer
Glass Filter Holder
Impingers with Absorbing Solutions
Thermocouple
Check Valve
Reverse-Type Pitot Tube
Stack Wall
Heated Area
Pitot Manometer
Ice Bath
Empty (Optional Knockout)
Silica Gel
5% HNO $_3$ /10% H $_2$ O $_2$
4% KMnO $_4$ /10% H $_2$ SO $_4$
4% KMnO $_4$ /10% H $_2$ SO $_4$
Thermocouples
Bypass Valve
Vacuum Gauge
Vacuum Line
Orifice
Main Valve
Air-Tight Pump
Dry Gas Meter

Figure 3. Recommended multiple metals sampling train configuration.

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TABLE 1. METALS DISTRIBUTION IN MULTIPLE METALS TRAIN

Metal	Percent in Front Half	
	First Test	Second Test
Antimony	64-100 (mixed)	> 97
Cadmium	100	100
Chromium	100	> 96
Copper	100	> 94
Lead	>98	> 99
Nickel	>81	> 85
Selenium	100	> 96
Thallium	---	100
Zinc	>98	> 97
Arsenic	0	100
Barium	0	> 85
Manganese	0	< 55
Mercury	< 2	< 5
Phosphorus	<15 (mixed)	> 95

TABLE 2. COMPARISON OF MERCURY RESULTS FOR MULTIPLE METALS VS. METHOD 101A TRAINS

Field Test	Multiple Metals Train		Method 101A Train		Relative Error of Metals Train vs. Method 101A Train* (%)
	Sample Identity	Flue Gas Hg Concentration (ug/dscm)	Sample Identity	Flue Gas Hg Concentration (ug/dscm)	
1	C1-3A	538.14	101A-3C	575.72	7.7
	C1-3B	615.38	101A-3D	495.2	
	C1-4A	7664.06	101A-4C	7259.44	5.3
	C1-4B	7102.98	101A-4D	6762.54	
2	TMC1-5	56.5	101A-5	86.5	-35.9
	TMC2-5	53.8	101B-5	85.7	
	TMC1-6	53.4	101A-6	73.2	-22.3
	TMC2-6	60.1	101B-6	73.0	

*Based on averages of two pairs of trains for single quad-train run.

TABLE 3. CALCULATED PRECISION OF SAMPLING AND ANALYSIS FOR METALS

	Relative Standard Deviation (%)[a]									
Run	Sb[b]	As[b]	Cd[c]	Cr[c]	Cu[c]	Pb[b]	P[c]	Se[b]	Tl[b]	Zn[c]
1	12.6	12.6	11.4	11.1	9.9	10.2	13.0	13.8	14.3	11.2
3	13.6	13.2	12.0	10.0	12.1	13.8	16.3	14.8	10.7	11.8
4	13.1	15.1	11.4	11.3	13.5	10.8	16.1	16.2	13.4	11.2
7	11.5	13.0	11.1	12.6	10.6	9.8	13.0	16.3	10.8	13.0
AVG	12.7	13.5	11.5	11.3	11.5	11.2	14.6	15.3	12.3	11.8

[a] Barium, beryllium, manganese, and nickel were detected but were below the quantifiable limit. Silver was not detected.
[b] Analysis by GFAAS.
[c] Analysis by ICAP within the working range.

TABLE 4. IN-STACK METHOD DETECTION LIMITS (ug/m^3) FOR TRAIN FRACTIONS USING ICAP AND AAS

Metal	Filter and Probe Rinse	Impingers 1-3	Impingers 4-5	Total Train
Antimony	7.7 (0.7)*	3.8 (0.4)*		11.5 (1.1)*
Arsenic	12.7 (0.3)*	6.4 (0.1)*		19.1 (0.4)*
Barium	0.5	0.3		0.8
Beryllium	0.07 (0.05)*	0.04 (0.03)*		0.11 (0.08)*
Cadmium	1.0 (0.02)*	0.5 (0.01)*		1.5 (0.3)*
Chromium	1.7 (0.2)*	0.8 (0.1)*		2.5 (0.3)*
Copper	1.4	0.7		2.1
Lead	10.1 (0.2)*	5.0 (0.1)*		15.1 (0.3)*
Manganese	0.5 (0.2)*	0.2 (0.1)*		0.7 (0.3)*
Mercury	0.05**	0.03**	0.03**	0.11**
Nickel	3.6	1.8		5.4
Phosphorus	18	9		27
Selenium	18 (0.5)*	9 (0.3)*		27 (0.8)*
Silver	1.7	0.9		2.6
Thallium	9.6 (0.2)*	4.8 (0.1)*		14.4 (0.3)*
Zinc	0.5	0.3		0.8

( )*Detection limit when analyzed by GFAAS.
**Detection limit when analyzed by CVAAS.

EVALUATION OF INCINERATOR ASH TESTING DATA -- A STATISTICALLY CORRECT APPROACH

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ABSTRACT

Several sets of EP Toxicity data for waste-to-energy plant ash are now in the public domain. These data sets usually include a few large values which cause the data transformation requirements of good statistical analysis and SW-846 guidelines to be triggered. Unfortunately, the SW-846 guidelines suggest transformation formulas which do not necessarily do the intended job for incinerator residues, ash. Methods for determining the need to transform the data, selecting correct transforms, and properly comparing the published results with regulatory thresholds are presented. This paper will help correctly classify ash so that RCRA compliance can take place.

INTRODUCTION

Under Federal and State regulations, the characteristics of a waste must be established and compared to regulatory thresholds to determine how the waste can be legally handled. After designing and conducting a proper sampling program, a set of representative samples and laboratory data results. With proper analysis, this data enables a meaningful conclusion to be drawn concerning the nature of the waste.

Sampling and analysis of a portion of the waste stream takes place because it is impractical to test all the waste leaving a plant. In addition to providing a point estimate of waste characteristics (the mean average of the data), the data also establishes an interval where, if the sampling were repeated a large number of times, we can state with a given certainty (confidence level) that the characteristic we would find is within the interval if we could test all the waste. Since the "true" value has an equal probability of residing anywhere within the calculated limits, if we calculate, say, an eighty percent confidence interval, we know that there is a 90 percent chance that the real value is less than the upper calculated limit (10 percent is below the interval, eighty percent is within it and 10 percent is above the interval).

Using the SW-846 (1) procedures, a waste is determined to be non-hazardous if the 80 percent upper confidence limit [UCL] is less than the standard. The UCL is calculated as:

$$UCL = x + t_{.,8}\, S/\sqrt{n}$$

where: $t_{.,8}$ is the 80 percent probability t-statistic for n-1 data points;

S is the standard deviation; and,

n is the number of data points in the analysis.

Interestingly, based on the operating characteristics for this statistical test (2), at the 80% probability level for the t-statistic, the risk of saying a waste passes when it really should not, is essentially equal to the risk of declaring a waste hazardous when it really is not. Consequently, it is important to resist the desire to pick a much higher confidence level; above about 87.5 percent probability level for the t-statistic, the risk of falsely classifying a waste rises very rapidly from about 10% to a near certainty.

In SW-846 (3), the EPA repeats the warning found in most statistics books: that establishing the characteristic of a waste and comparison of test results to standards using statistical procedures based on the normal distribution are only valid if the data being analyzed is reasonably normally distributed. SW-846 recommends that the data be transformed if the variance (standard deviation squared) exceeds the mean and, depending on the amount of excess, that the data be "normalized" by using either the arcsine transform (mean is less than the variance) or square root transform (mean is about equal to the variance).

After transforming the data, the UCL is calculated for the transformed data and the result compared to the regulatory limit after it is transformed the same way the data was. Alternatively, the UCL can be inversed back into the untransformed data plane by reversing the mathematics of the transformation process. For example, if the data is square root transformed, the UCL can be inversed by squaring. For lead, the square root of the standard (5 mg/l) is 2.24; a transformed plane UCL of $1.70\sqrt{mg/l}$ inverses to 2.89 mg/l.

The arcsine transform applies [illegible]a that is either symmetric, but not bell shaped, or has a tail of small values and peaks near the high end of the data set (4). As a result, the application of the arcsine transform to ash data, which tends to have a large value tail, is questionable. In fact, the arcsine transform, in the author's experience, does not usually normalize incinerator ash data.

In addition, the criterion used to identify non-normal data sets in SW-846 (a comparison of the mean to the variance) is not as powerful as is readily available and no guidance is offered in SW-846 to suggest that the transformed data should be checked for normalcy. This means false conclusions may be reached. There is a particularly great risk of reaching false conclusions with incinerator ash data because long tails (some big values) are inevitable. At the very least, identifying the need for a transformation and selecting the right one will be the result of happenstance rather than design as will reaching the correct regulatory conclusion.

This paper provides some practical advice on dealing with ash data. While theoretical justifications exist for points discussed, only "bottom lines" are provided to facilitate use of our findings by the general practitioner. To keep the paper length within reason, only combined (bottom and air pollution control) ash lead data will be discussed in the following. Our experience indicates that all metals except lead and cadmium are typically sufficiently far below the regulatory threshold so that the methods presented herein are of interest for reasons of purity rather than practicality. Also, cadmium is normally distributed; on the other hand, lead is usually not, so that interesting, real examples are found for lead. The same statistical treatment is needed for air pollution control (fly) ash; the mean values are just higher. The techniques are applicable, however, to any data being analyzed.

The problems covered, using real public data sets, include:

- Is the data normally distributed? How do you know? How can you normalize non-normal data?
- What do you do with "below detection limit" data?
- What do you do with extreme values?
- What do you do with data taken over the long term?

DATA NORMALCY

Mathematical Approach

In addition to the well-known first and second moments of a data distribution -- the mean[1] and standard deviation[2] -- the next two moments -- skewness[3] and kurtosis[4] -- are frequently calculated by readily available statistical analysis programs such as SPSS-P20 statistics program implemented on our DEC PRO 350 and 380 computers.

If the underlying distribution is normal, then the skewness and kurtosis should ideally be zero and, for the stated probability, are certainly less than the values tabulated in Table 1. Consequently, an important check to make before calculating the UCL is that the skewness and kurtosis are characteristic of normal distributions. Otherwise, the UCL for a given numerical value of the t-statistic has an unknown significance level and a false classification can result when the calculated UCL is compared to the regulatory threshold.

Table 2 shows several calculated statistical parameters for publicly available combined ash lead data from Marion County, OR (EP Toxicity)(5), Commerce, CA (EP Toxicity and California Assessment Manual [CAM] Toxicity)(6), and a composite of the EP Toxicity data published for 23 plants (7 - 22), including those previously mentioned. Normalized data sets are indicated by the underscored transform description. The preferred result, based on using no transform if the data is already normal and the transform which gets closest to theoretical zero values for skewness and kurtosis if several transforms appear acceptable, is also underscored.

In addition to the transforms recommended in SW-846, the natural logarithm transform is computed because it applies where the mean is about equal to the standard deviation (23).

Only the Commerce Lead EP Toxicity data is normally distributed without transformation or some other treatment. SW-846 guidelines indicate, however, that a transform is needed. The variance is 1.61 which is about equal to the mean of 1.63 so that the square root transform is indicated.

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[1] mean = $\bar{x} = 1/N \, \Sigma \, x_i$ where x_i is the i^{th} measurement, there are N measurements and Σ denotes the sum of all x_i from 1 to N

[2] standard deviation = $S = 1/N-1[\Sigma(x-\bar{x})^2]$

[3] skewness = $\sqrt{N} \, \Sigma \, (x_i-\bar{x})^3/[\Sigma(x_i-\bar{x})]^{3/2}$

[4] kurtosis = $N \, \Sigma(x_i-\bar{x})^4/[\Sigma(x_i-\bar{x})^2]^2$

The Marion County and Commerce data has been transformed by others, using the arcsine transform, for regulatory purposes. This transformation was accomplished by first dividing the data by the largest measured value (the arcsine is only defined between -1 and 1) and then taking the arcsine of the square root of the dividend. The arcsine transformed data for Marion County is not normal. This means that when the upper confidence limit was calculated and compared to the regulatory limit, the probability that the characteristic is less than the standard is unknown.

It is extremely important to realize that the t-statistic based confidence interval calculation applies to reasonably normal data. As a result, if non-normal data is involved, the calculation can be performed but the probability associated with the UCL is unknown for the actual data distribution.

Fortunately, the t-statistic based confidence interval is reasonably insensitive to small departures from normalcy. As a result, if the data cannot be completely normalized using transforms, but the skewness and/or kurtosis can be made reasonably close to the values tabulated in Table 1, no material problem will likely result. If the miss is severe, however, the conclusion reached when comparing to regulatory thresholds can be wrong.

Returning to Table 2, it is clear that direct application of the confidence interval calculation without appropriate transformation would result in declaring the Marion County combined ash (and combined ash taken from the published composite of plants in the United States) hazardous. Basing the conclusion on this calculation would be wrong! The data is not normally distributed. If the data is transformed by taking the log of the parameter, this set can be normalized. The properly handled data indicates the combined ash is not hazardous. In fact, all the sets in Table 2 which require transformation are normalized by either the log or square root transformation. This result is not surprising since these transformations apply to data sets with long tails.

Graphical Approach

Another approach to judging data normalcy is to sequence the data from smallest to largest and plot the data value against the normal deviate (Ƶ-score) associated with the median rank of each sequenced point. Since the Ƶ-score is the number of standard deviations a given cumulative probability is found above or below zero, this is the same as plotting the data on probability paper.

A Ƶ-Score plot can be readily prepared within a computerized statistical analysis package as long as data can be transformed and scatterplots prepared.

For a sequenced data set containing N data points, the median rank of each data point can be approximated for the i^{th} point using: $P = (i - 0.3)/(N + 0.4)$. The normal deviate can then be calculated using the following approximation (24):

for $P \leq 0.5$:
$T = \mathrm{SQRT}\,[-2.0 \times \ln(P)]$
$Ƶ = T - (2.30753 + 0.27061T)/(1 + 0.99229T + 0.04481T^2)$

and, for $P > 0.5$:
$T = \mathrm{SQRT}\,[-2.0 \times \ln(1-P)]$
$Ƶ = (2.30753 + 0.27061T)/(1 + 0.99229T + 0.04481T^2) - T$

When the data is properly normalized, the plot of the transformed data versus Ƶ will be reasonably close to a straight line. Data sets clearly described by curves or two different lines are not normally distributed. From the plot, the mean can be found at Ƶ = 0 and the standard deviation can be taken as one third the difference between the values at $Ƶ = -1.5$ and $Ƶ = 1.5$ if a full set of descriptive statistics have not been run.

Figures 1 through 3 show for Marion County, Commerce and "All Facilities": (*a*) the untransformed data; (*b*) the natural log transformed data; (*c*) the square root transformed data; and (*d*) the arcsine transformed data.

For Marion County, the natural log transformed data is normal: no other transform is.

For Commerce, the untransformed data is normal as is the square root and arcsine transformed data. Since no transform is required for the data, none should be used. Also, if the arcsine transform is done in the conventional manner, the divisor is less than 5. Thus, the transformed regulatory limit is undefined. The comparison to the regulatory limit cannot be done in the transformed plane as recommended by SW-846 (25).

The Commerce CAM (California Assessment Manual -- STLC or WET Method) lead toxicity data is normalized by the square root transform as is the "All Facilities" data set.

Proper conclusions require that the data be normalized before applying normal statistics. This can be adhered to by checking normalcy, before drawing conclusions, using graphically computerized techniques.

BELOW DETECTION LIMIT DATA

Many reports contain "below detection limit" results. This data should not be excluded from the analysis, but what value to use is problematic. From a regulatory perspective, there is a certain logic to arguing that since the metal might be present very near to the detection limit, using the detection limit is conservative. Facility operators might want to substitute a value of zero because this will reduce the average value.

Neither of these approaches is proper because the physics of the situation argues for some of each element to be present and that presence to be distributed between zero and the detection limit. Exclusively locating at the upper or lower extremes is physically unreasonable. To deal with this situation, some statisticians recommend using half the detection limit. An alternative approach, to me intuitively satisfying, is based on an extension of the confidence interval concept. Since the mean value for the entire waste stream is equally likely to be anywhere within the confidence interval estimated from the sample, if a uniform probability distribution is assumed for the actual data value between zero and the detection limit, then a distribution of values can be assigned to "below detection limit" data rather than a single point estimate. Such an approach becomes particularly important if skewness and kurtosis are to be used to determine normalcy -- the below detection limit values are at the small end of the distribution and can result in creating an apparently non-normal data set when, in reality, the data is normal.

Finally, the data can be "Windsorized" as described in the next section.

My experience is that either a random uniform assignment of Below Detection Limit data or using half the detection limit works equally well from a practical standpoint. The uniform distribution assignment may be preferred by many on intuitive grounds.

EXTREME VALUE DATA

Extreme value data (say more than 3 or 4 standard deviations away from the mean) that cannot be eliminated because of a discernible sampling or laboratory mistake is usually an indication of a statistical aberration or a very real and interesting occurrence. Consequently, it is generally bad practice to eliminate otherwise apparently valid data on distribution grounds alone.

When looking at ash data, eliminating apparent "outliers" from the analysis on statistical grounds is very dangerous because a value much in excess of the average is not necessarily an outlier. Despite the best sampling program design and execution and laboratory analysis, a large lead value can result, for example, if the waste being burned contains an unusual number of batteries, coated cans or if the sampler has the misfortune to grab a sample with a lead wine bottle foil. As a result, valid lead data sets are likely to be tailed to the right (large values) for elements that are found in specific waste components and which may not be uniformly dispersed. This is an inherent consequence of the heterogeneous nature of the waste being burned.

When dealing with a skewed distribution, it is mandatory that the data be transformed to as normal a set as possible before applying any rejection criteria. If, after transformation, the set plots and tests as normal, no further consideration is needed. If, on the other hand, an end data point appears to remain outside the data set, it can be tested using generally published rejection criteria (26). Returning to Figure 1(b), it appears that the largest lead value (3.045 log transformed) might be an outlier. Using the log transformed mean (0.860) and standard deviation (0.837), it is clear that the extreme value is only 2.6 standard deviations away from the mean. This is too close to be considered an outlier using any rejection criteria and must remain in the data set for analysis. This is true even though, for untransformed data, the largest value is 3.84 standard deviations away from the mean and has a very low probability of occurring -- less than 0.005%.

If the largest lead data point is deleted from the Marion County data, the UCL in original units becomes 2.92 mg/l as compared to the natural log transform UCL for the complete data set of 3.08 mg/l. The skewness and kurtosis become 0.32 and 0.78 -- normal values.

Another way of handling extreme value data is to "Windsorize" the data. In this process, the lowest value is replaced by the second low and the highest value is replaced by the second high. This process can be repeated until a normal data set results. If the Marion County data is "Windsorized", the UCL for lead becomes 3.10 mg/l in original units.

For the Marion County data, all three approaches result in essentially the same original unit UCL. This is strictly a fortuitous occurrence and should not be counted on.

To this point, I have talked about extremely large values. Small values can also be outliers. Both ends of the distribution need to be checked. In the case of ash characterization data, the likelihood is an unusual number of "below detection limit" values may occur. While of little concern for metals which are well below the standards, for lead where large extreme values naturally occur, one should not become too enthusiastic about finding a transform that "straightens" out the big values and skews the small values. Under this circumstance, the calculated UCL will still have an unknown probability. Both mathematical and graphical approaches to checking normalcy will uncover non-normal small values. Transformation, deletion and Windsorizing will work on small as well as large outliers.

COMBINING DATA

Characterization of an incinerator ash stream is frequently an ongoing process. After an initial characterization study, there may be an ongoing monitoring program. The question naturally arises whether each period's data should be treated separately or combined with the original data set before calculating the UCL and comparing the results to the regulatory standard.

If it can be shown that the new and old data are the same, then the data can be properly combined for comparison to the standard. If the data cannot be shown to be the same, then the new data should be separately analyzed and compared to the standard. Even with the same mean and standard deviation, the UCL will be greater for a small data set than for a larger data set. There is an incentive to combine data when this can be properly done.

One of the most powerful ways to compare two data sets is to perform a one-way Analysis of Variance [ANOVA] of data. Like the UCL calculation, this test assumes that the data is normally distributed. Consequently, it is imperative that the data first be checked for normalcy.

To begin the test, the old and new data are assumed to come from the same underlying distribution and all the data is transformed to a reasonably normal distribution using the transform that worked for the old data. A probability plot will immediately disclose a severe case of the data coming from two different distributions because all the new data will be described by a different straight line than the original data! After the combined data is normalized, then the sets of transformed data (original and new) can be compared by performing ANOVA. This is a commonly available statistical procedure which essentially asks whether the confidence intervals for the new and old data sets overlap at a specified probability. Table 3 presents the ANOVA results for CAM Toxicity lead between different sampling periods at the Commerce Plant. Since it is unlikely that the data sets are different, they are properly combined for comparison to standards as was done earlier in this paper.

If all the published EP Toxicity data of which we are aware is combined and treated as if the combined ash from each facility were coming from a single plant, the as-published data is not normally distributed, but the log transformed data has a nearly normal skewness and the Ƶ-score plot of the sequenced data plot is reasonably straight. Giving each piece of combined ash data equal weight leads to the conclusion that the nationwide upper confidence limit ash characteristic for lead is 0.683 (natural log transformed plane) or 1.97 mg/l in original units compared to the regulatory threshold of 5 mg/l.

If we treat each plant as having equal weight (e.g. removing the bias introduced by testing one plant more than another), then the square root transformed data is nearly normal and the Ƶ-score plot of the transformed data is reasonably straight. The UCL ash characteristic for lead is 2.22 in the transformed plane or 4.92 mg/l in original units.

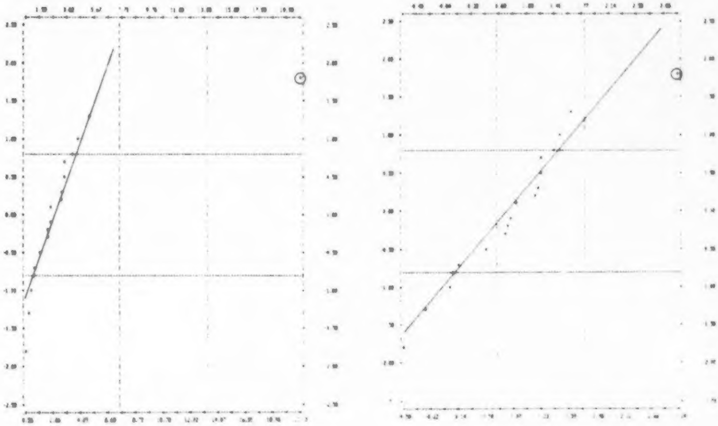
BOTTOM LINE

Incinerator ash lead content characterization data is almost never normally distributed and other characteristics may also be non-normal due to the inherently heterogeneous nature of the waste being burned. Because there are physical reasons why ash toxicity data will be skewed, it is important that anyone analyzing the data be aware of this nature.

In order to correctly determine the characteristic of the waste, it is necessary to establish that the data is normally distributed. If not, transform the data into a normal distribution before doing calculations and comparing to regulatory limits. Normalizing transforms should also be performed prior to deciding that a data point is an outlier or deciding whether data should be combined or separately analyzed.

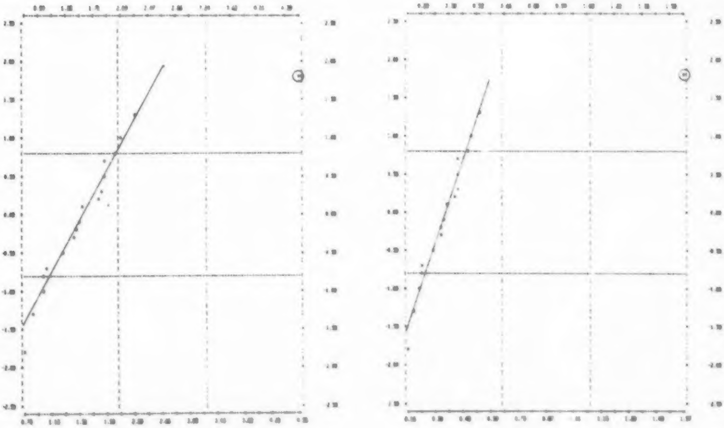
The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

FIGURE 1. Z-SCORE (NORMAL PROBABILITY) PLOTS OF THE MARION COUNTY COMBINED ASH EP TOXICITY LEAD DATA



(a) Untransformed

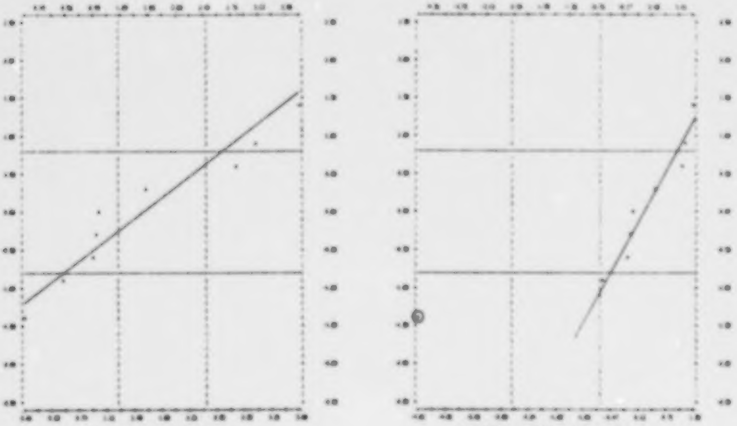
(b) Natural Log Transform



(c) Square Root Transform

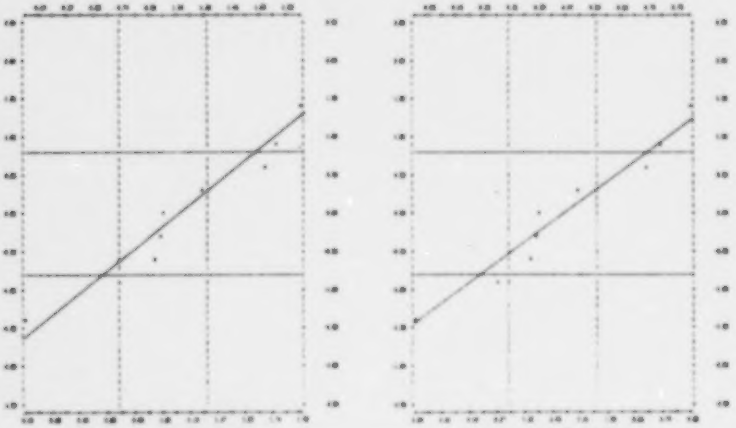
(d) Arcsine Transform

FIGURE 2. Z-SCORE (NORMAL PROBABILITY) PLOTS OF COMBINED ASH EP TOXICITY LEAD RESULTS FROM COMMERCE, CA.



(a) Untransformed

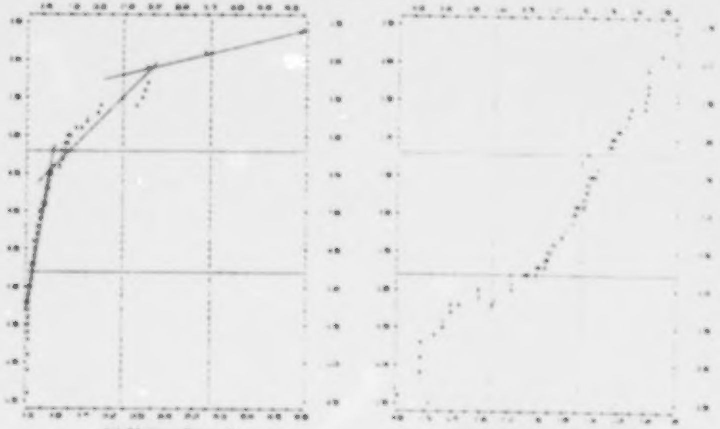
(b) Natural Log Transform



(c) Square Root Transform

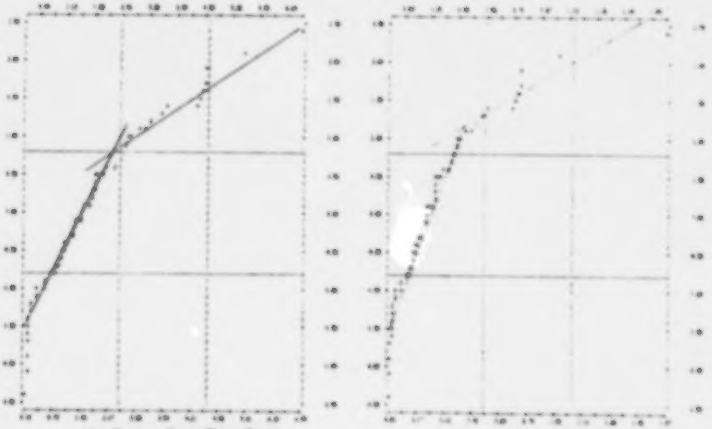
(d) Arcsine Transform

FIGURE 3. Z-SCORE (NORMAL PROBABILITY) PLOTS OF EP TOXICITY LEAD DATA FOR MANY PLANTS



(a) Untransformed

(b) Natural Log Transform



(c) Square Root Transform

(d) Arcsine Transform

TABLE 1. LIMIT VALUES OF SKEWNESS AND KURTOSIS FOR NORMAL DISTRIBUTIONS*

PERCENT SIG. LEVEL	NUMBER OF DATA POINTS										
	5	10	15	20	25	30	35	40	50	75	100
					SKEWNESS						
1	1.34	1.31	1.20	1.11	1.06	0.98	0.92	0.87	0.79		
5	1.05	0.92	0.84	0.79	0.71	0.66	0.62	0.59	0.53		
					KURTOSIS						
1	3.11	4.83	5.06	5.23	5.00				4.88	4.59	4.39
5	2.89	3.85	4.07	4.15	4.00				3.99	3.87	3.77

* Abstracted from Tables 5 and 6 appearing in ASTM Standard E178

TABLE 2. SUMMARY STATISTICS FOR TOXICITY TEST LEAD FROM SEVERAL ASH CHARACTERIZATION DATA SETS IN THE PUBLIC DOMAIN

"OUTLIER" TREATMENT	NORMALIZING TRANSFORMATION	NO. POINTS	MEAN	STD. DEV	SKEW-NESS	KUR-TOSIS	TRANSFORM STANDARD	TRANSFORM UCL	INVERSED UCL
				MARION COUNTY, OR -- EP TOXICITY					
NONE	NONE	18	3.51	4.26	3.67	14.59	5.00	4.86	4.94
	SQUARE ROOT	18	1.48	0.85	2.45	8.23	2.24	1.95	3.79
	NATURAL LOG[a]	18	0.86	0.86	0.61	1.64	1.61	1.12	3.08[b]
	ARCSINE √X/[illegible]	18	0.41	0.31	3.43	13.30	0.51	0.51	4.96
WINSORIZED	NONE	18	2.64	1.45	0.34	0.93	5.00	3.10	3.10
	SQUARE ROOT	18	1.56	0.46	0.06	1.15	2.24	1.70	2.91
	NATURAL LOG	18	0.80	0.64	0.46	1.05	1.61	1.00	2.72
	ARCSINE √X/5.16	18	0.65	0.36	0.71	0.65	1.39	0.96	3.40
LARGEST VALUE DELETED	NONE	17	2.48	1.26	0.32	0.78	5.00	2.92	2.92
	SQUARE ROOT	17	1.51	0.43	0.13	1.01	2.24	1.66	2.74
	NATURAL LOG	17	0.73	0.60	0.59	0.66	1.61	0.94	2.57
	ARCSINE √X/[illegible]	17	0.60	0.33	0.65	0.38	1.39	0.91	3.20
				COMMERCE, CA -- EP TOXICITY					
NONE	NONE	9	1.63	1.27	0.55	1.17	5.00	2.22	2.22
	SQUARE ROOT	9	1.16	0.57	0.30	0.66	2.24	1.43	2.03
	NATURAL LOG	9	-0.16	1.70	2.25	5.80	1.61	0.67	1.96
	ARCSINE √X/[illegible]	9	0.75	0.47	0.46	0.18	UNDEFINED	0.97	2.50
	ARCSINE √X/[illegible]	9	0.47	0.25	0.13	0.47	0.41	0.50	2.25
				COMMERCE, CA -- CAM TOXICITY					
NONE	NONE	33	13.71	10.47	1.50	2.85	5.00	16.10	16.10
	SQUARE ROOT	33	3.45	1.37	0.40	0.20	2.24	3.76	14.15
	NATURAL LOG	33	2.20	0.90	0.76	0.35	1.61	2.51	12.24
	ARCSINE √X/48.36	33	0.56	0.20	1.47	3.82	0.33	0.63	16.57
	ARCSINE √X/[illegible]	33	0.37	0.16	0.67	0.81	0.23	0.41	15.09
				"ALL" PUBLISHED FACILITIES -- EP TOXICITY					
NONE	NONE	80	4.60	7.41	3.48	15.43	5.00	5.70	5.70
	SQUARE ROOT	80	1.73	1.30	1.42	2.77	2.24	1.91	3.64
	NATURAL LOG	80	0.42	1.09	0.80	0.32	1.61	0.88	1.87
	ARCSINE √X/[illegible]	80	0.27	0.24	2.56	10.25	0.33	0.30	4.34
PROP. WEIGHT	NONE	80	3.48	6.09	2.93	12.44	5.00	6.62	6.62
	SQUARE ROOT	80	2.05	1.23	0.97	1.54	2.24	2.22	4.92
	NATURAL LOG	80	0.97	1.37	1.27	1.93	1.61	1.18	3.27
	ARCSINE √X/[illegible]	80	0.34	0.22	1.99	6.08	0.33	0.34	5.43

[a] UNDERSCORE INDICATES NORMAL TRANSFORMED DATA

[b] UNDERSCORE INDICATES PREFERRED DATA HANDLING UCL

TABLE 3. ANOVA BETWEEN WET (CAM) TEST LEAD TAKEN 10 DIFFERENT MONTHS

SQUARE ROOT OF LEAD BY TEST MONTH

SOURCE OF VARIATION	SUM OF SQUARES	DF	MEAN SQUARE	F	SIGNIFICANCE OF F
MONTH	30.723	10	3.072	2.306	0.049
RESIDUAL	29.315	22	1.333		
TOTAL	60.038	32	1.876		

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COORDINATED WASTE, ASH, AND EMISSIONS SAMPLING AT THE COMMERCE WASTE-TO-ENERGY FACILITY

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ABSTRACT

The Commerce waste-to-energy facility is a 350 ton/day commercial refuse incinerator which generates 11 megawatts of electricity and is equipped with ammonia injection, a dry scrubber and a baghouse for emissions control. The California Waste Management Board, in cooperation with other state agencies, and the County Sanitation Districts of Los Angeles County performed extensive testing of two different waste streams (residential and commercial), ash, and emissions from the facility in July, 1988. Emissions testing was performed simultaneously by two different test teams. Preliminary findings show that residential wastes tended to be higher in certain heavy metals than the commercial wastes, but that the ash results were variable. Emissions of dioxins and furans, PAHs, and metals were similar to that measured in other similarly equipped facilities.

BACKGROUND

California is facing decreasing landfill capacity and a growing awareness of the possible adverse environmental effects associated with landfills. In response to these concerns, communities across the state have considered alternate disposal methods including waste incineration. Several facilities were proposed and there are now three operating waste-to-energy (WTE) plants in the state; however, many other proposed projects were rejected by the public due to environmental and health concerns.

In response to requests by the California Waste Management Board (CWMB) for funding to study the impacts of the technology, the state legislature allocated one million dollars for a demonstration program. In June of 1987, the CWMB convened a technical advisory committee (TAC) comprised of state agencies involved in the permitting of WTE facilities who could provide insights into information gaps that needed to be filled. The agencies are: the California Air Resources Board, the California Department of Health Services, the State Water Resources Control Board, the Central Valley Regional Water Quality Control Board, the California Energy Commission, the County Sanitation Districts of Los Angeles County, and the South Coast Air Quality Management District. The TAC recommended focussing on the ability of the technology to comply with the environmental and public health regulations of the state, specifying the need to characterize the waste, ash, and air emissions of the facility.

The TAC developed sampling and analytical protocols for the three components and selected the Commerce WTE facility as the test site. The Commerce plant was chosen as it was the first commercial WTE plant in operation and it was felt that its relatively small size (350 TPD) would allow greater flexibility. Actual testing began in July of 1988 and ran for three weeks.

The WTE Demonstration Program was designed to examine plant operation at opposite ends of the waste spectrum. At one end was the plant's regularly permitted waste stream composed of ~95% commercial waste and 5% residential; at the other, the committee specified use of a mixed waste composed of 60% residential and 40% commercial. By sampling these two waste streams and the resulting ash and emissions, it was hoped to obtain an understanding of how fuel characteristics and plant operation affected the plant's ability to meet environmental regulatory limits.

PLANT DESCRIPTION

The Commerce WTE plant is designed to burn between 300 and 350 tons of high grade commercial waste a day. The waste stream is generated within the City of Commerce which, as the name indicates, is a city consisting largely of commercial and industrial establishments. Its residential population is around 15,000 but its workday population swells to 40,000. Analyses of the waste done in 1982 yielded an estimated heat content of 6250 BTU/lb.

Waste is stored in a pit with a storage capacity of 1200 tons. The boiler consists of a Foster Wheeler mass-burn water-wall boiler designed to deliver 115,00 lb/hr of steam at 650 psig and 750 degrees F. The grate is a three-stage reciprocating grate designed by Detroit Stoker and has an area of 490 ft^2, a gross heat release rate of 336,000 BTU/ft^2-hr, and a 18 inch drop between sections.

Air pollution control equipment at the plant consists of ammonia injection for control of oxides of nitrogen, a dry-scrubber with lime-slurry injection followed by a dry venturi injected with an agglomerating agent, and a baghouse for control of acid gases and particulate matter.

TEST PROGRAM

Testing on the mixed residential waste stream began on Monday, July 18. (Figure 1 gives the test schedule for the three weeks). The plant began burning the new waste that morning and experienced some difficulty maintaining combustion. Due to the operational difficulties, the only emissions test run was a dioxin train, no ash or waste sampling had been planned for that day. Carbon monoxide levels correlate with the poor combustion conditions experienced at that time. By Tuesday, the plant operation had stabilized and the first ash and waste sampling was started. A total of seven ash samples and four waste samples were taken from Tuesday to Friday for the residential week. The emissions testing, which were more extensive, ran from Monday through Saturday and consisted of parallel sampling done by two teams at the boiler outlet and the stack.

Testing on the commercial waste stream was done over a two week schedule. A total of seven ash samples and six waste samples were taken during that first week on commercial waste. A larger number of waste samples were taken for the commercial week as it seemed reasonable to assume that there would be more variability through the day. While the homogeneity within a truckload may be greater, there could be more variability from truck to truck, thereby increasing the likelihood that a single truckload might influence operating conditions significantly.

During the emission testing on Thursday the 28th, the filter from the metals train had noticeable reddish-brown deposits. Upon later investigation, it was discovered that one bag in the baghouse had fallen from its mounting. The lab analysis of the sample indicated an increase in emissions. (See page 13)

During the third week, only chromium and formaldehyde emission tests were conducted.

WASTE SAMPLING

The waste sampling schedule for the mixed week consisted of grapple loads taken from the pit at one hour intervals throughout the day, Tuesday through Friday, for a total of four samples. For the commercial week, sampling was split between the morning and afternoon on three days for a total of six samples. As mentioned earlier, this was done to give a finer resolution to the waste variation with time.

60% Residential / 40% Commercial

95% Commercial / 5% Residential

Activity	MON 7/18	TUE 7/19	WED 7/20	THUˆ 7/21	FRI 7/22	SAT 7/23	SUN 7/24	MON 7/25	TUE 7/26	WED 7/27	THU 7/28	FRI 7/29	SAT 7/30	SUN 7/31	MON 8/01	TUE 8/02	WED 8/03	THU 8/04	FRI 8/05
Ash Sample	/ /	*AM #	AM PM	AM PM	AM PM	/ /	/ /	/ /	*AM #	AM PM	AM PM	AM PM	/ /	/ /	/ /	/ /	/ /	/ /	/ /
Emission Test	DT	M5 TM #	*DT #	RT M5 *TM	DT TM N	RT M5 N	/ /	/ RT N	# TM	DT TM *M5	DT TM	RT M5	/ /	/ /	M5 DT	CHR RT	CHR CHR F	DT M5 CHR	TM
Waste Sample	/ /	D A Y	*D A Y	*D A Y	D A Y	/ /	/ /	/ /	AM PM	AM PM	AM PM	/ /	/ /	/ /	/ /	/ /	/ /	/ /	/ /

ˆ switched to commercial from 5:30 P.M. to 5:30 A.M.
/ no sample taken
* late
# sample missed

DT = DIOXIN
M5 = METHOD
TM = METALS
RT = PAH/PCB
CHR = CHROMIUM
N = NITROSAMINES
F = FORMALDEHYDE

FIGURE 1: Commerce WTE Demonstration Calendar

The waste samples were transported to the Puente Hills Landfill for sorting into the following components:

- Paper
    - mixed
    - newsprint
    - corrugated
- Plastic
    - film
    - rigid
    - PET
    - other
- Wood/lumber
- Yard wastes
- Textiles
- Food
- Other organic
    - (rubber & leather)
- Ferrous
- Non-ferrous
- Aluminum
- Batteries
- Wire
- Glass
- Other inorganic

Sorted sample size ranged from 865 to 1118 pounds. Sub-samples were prepared for laboratory analysis. Proximate and elemental analyses were performed for each component and ultimate analyses for the organic fraction. Organic analyses were performed on composite samples. (See Tables 1 and 2)

RESULTS OF WASTE ANALYSIS

The waste characterization analyses yielded the following interesting differences in the two waste streams. The commercial waste was quite a bit higher in paper, textiles, and wood; the residential waste higher in yard waste and somewhat higher in glass. The glass component in each waste stream was lower than in other studies conducted nationwide. Surprisingly enough the aluminum content was quite low in both, and there was little difference between the wastes in the plastic and ferrous portions.

Waste Composition

	Comm.	Res.		Comm.	Res.
Paper	40.0%	30.9%	Glass	2.9%	3.8%
Plastic	10.8%	9.1%	Aluminum	0.4%	0.6%
Yard	3.1%	26.9%	Ferrous	6.7%	5.9%
Wood	12.9%	5.3%	Other Org	5.4%	3.0%
Textile	9.4%	4.2%	Other Inorg	4.1%	4.8%

Within the paper component, newsprint was higher in the residential waste while corrugated and mixed were higher in the commercial waste. Within the plastic, PET was surprisingly consistent while the film plastic showed quite a bit of variation from day to day, being overall higher in the commercial waste.

In examining the elemental analyses, several interesting observations can be made. The average lead concentration for the residential week was 123.1 milligrams per kilogram of waste (range: 19.3 to 349.1 mg/kg). Of that,

Table I Laboratory Analyses Performed on Each Solid Waste Sample

	Waste Component															
Analysis	Corrugated	News	Mixed Paper	Plastic[a]	Textiles	Yard	Wood	Food	Other Org.	Ferr	Non-Ferr	Al	Glass	Other Inorg	Composite	Total
27 Trace Metals	1	1	1	1	1	1	1	1	1	1	1	1	1	1		14
Proximate																
- Moisture	1	1	1	1	1	1	1	1	1	1	1	1	1	1		14
- Volatiles	1	1	1	1	1	1	1	1	1	1	1	1	1	1		14
- Ash	1	1	1	1	1	1	1	1	1	1	1	1	1	1		14
Ultimate																
- C,H	1	1	1	1	1	1	1	1	1				1	1		11
- N	1	1	1	1	1	1	1	1	1				1	1		11
- S	1	1	1	1	1	1	1	1	1				1	1		11
- Cl	1	1	1	1	1	1	1	1	1				1	1		11
PAH															1	
PCB															1	
PCDD/PCDF															1[b]	

[a] In addition to the analyses of the combined plastic category, one representative sample of film, of rigid, of PET, and of other plastic was analyzed for chlorine and metals.

[b] The composites were formed from every two consecutive samples of solid waste delivered to the landfill

Table 2 Methods for Laboratory Analyses of Solid Waste

Analysis or Procedure	Method
Sample Preparation	ASTM E829
Organic	
PCDD/PCDF	EPA 8290
PAH	EPA 8310
PCB	EPA 8080
Ultimate	
C,H	ASTM E777
N	ASTM E778
S	ASTM E775
Cl	ASTM E776
Proximate	
Total Moisture	ASTM E949
Residual Moisture	ASTM E790
Volatile Matter	ASTM E897
Ash	ASTM E830
Trace Metals	ASTM E926 Preparation Method B AA
As	•
Be	•
Cd	•
Cr	•
Ni	•
Hg	•
Zn	•
Cu	•
Co	•
B	•
P	•
Mo	•
Se	•
Mn	•
Sb	•
Ba	•
Bi	•
Mg	•
Al	•
K	•
Pb	•
Ca	•
Fe	•
Si	•
V	•
Ag	•
Tl	•

nearly 45% was contributed by the ferrous component alone which was present in the waste sample at an average concentration of 56 mg per kg of waste. (The ferrous component concentration of the different samples taken that week ranged from .33 to 218 mg/kg.) The average zinc concentration in the residential waste was 226.2 mg/kg (range: 91.9 to 620.9). On average, 53% of the week's zinc came from yard wastes due to a single day with the exceptionally high value of 437 mg/kg, (average of 119.1 mg/kg, range: 2.1 to 437 mg/kg).

For the commercial week, the greatest contributor of lead was again the ferrous component providing 35% of the lead in the waste samples (the average contribution was 29.8 mg/kg, the range: 0.2 to 158 mg/kg). The overall concentration of lead in the commercial waste was 85 mg/kg (range: 28.4 to 180) compared to the higher residential value of 123.1. The commercial zinc was dominated by the contribution of the other-organic component at 40%, with an average concentration of 126 mg/kg, also lower than the residential waste. (The other-organic category primarily contains rubber, leather, and used diapers, although there were no diapers in the commercial waste)

ASH SAMPLING

Ash sampling was performed on four days for each waste stream. Morning and afternoon samples were taken, except for the first day of each week when only morning samples were taken, bringing the total number of samples for each week to seven. Grab samples were obtained from the end of the conveyor belt with a front-end loader at thirty minute intervals for a collection period of thirty seconds. Over the course of a sampling period, nine grab samples were taken for compositing into the final sample. Large, extraneous objects were removed from the pile and the ash was passed through a 2-inch screen. (Oversized objects were subjected to an EPA recommended integrity test.) The remaining ash was riffled into grid forms, halved, redistributed into the grids and samples selected on the basis of a random number generator.

RESULTS OF ASH ANALYSIS

The ash samples were analyzed under the California waste extraction test (WET) for 21 metals, certain organics, and other parameters listed in Table 3. Initial analyses were done on 40 samples spread throughout the two weeks. Based on the results of those analyses, further analyses were proposed for the WET, federal extraction procedure toxicity test (EP Tox), loss on ignition tests, and total organic carbon.

Results from the first round are highly variable and it is difficult to draw conclusions; however some data are of interest. Concentrations of lead in the residential ash for the soluble threshold limit concentration (STLC, limit = 5 mg/l) ranged from 2.2 to 34 mg/l with the average being 13.9 mg/l. The rate of failure of the STLC limit for the individual samples was 70%. Lead concentrations in the commercial ash were 1.4 to 31 mg/l, average 17.1 mg/l, failure rate 95%. For zinc (STLC limit 250 mg/l) in the residential ash, the range was 97 to 200 mg/l, average at 38.6 mg/l (under the limit). The range

TABLE 3

WASTE-TO-ENERGY DEMONSTRATION PROJECT – ASH TESTING

Analyses:

Constituents and Parameters:

		Total concentrations	California Waste Extraction Test – citrate buffer extraction	California Waste Extraction Test – deionized water extraction
Metals –	ICAP Scan for the following 17 metals:	•	•	•
	Aluminum			
	Antimony			
	Barium			
	Beryllium			
	Cadmium			
	Chromium (Total)			
	Cobalt			
	Copper			
	Iron			
	Lead			
	Manganese			
	Molybdenum			
	Nickel			
	Silver			
	Thallium			
	Vanadium			
	Zinc			
	Arsenic'	•	•	•
	Chromium (VI)'	•		
	Mercury'	•	•	•
	Selenium'	•	•	•
Organics –	Chlorinated dibenzodioxins & dibenzofurans*	•		•
	Polynuclear aromatic hydrocarbons*	•		•
Miscellaneous –	Ammonia (-ium)	•		
	Chloride	•		
	Fluoride		•	•
	Sulfate	•		
	Specific conductivity	•		
	Total dissolved solids	•		
	Total organic carbon			•
	pH	•		•
Fish Bioassay –	96 hr. LC50 (250, 500, 750 mg/l screening)*			•

---

* Run on every other sample.

' ICAP Scan may not be used for these metals.

for zinc in the commercial ash was 84 to 280 mg/l, the average 160 mg/l, and the failure rate 18%.

It is of interest that the soluble concentration of lead in the ash from commercial waste was higher than in the residential ash, while the waste analysis yielded the opposite; residential lead being, on average, higher than commercial. This holds for the total threshold limit concentration (TTLC) for lead with the residential at 1700 mg/kg and commercial at 2500 mg/kg. The zinc concentrations were consistently higher in both the waste and ash for the residential samples than for the commercial.

DIOXINS

Dioxin/furan analysis indicated that the hepta and octa congeners are predominant in both the combined and fly ashes (see Table 4). The distribution of furans is not so clear cut, but the penta and hexa forms seem favored.

AIR EMISSION TESTING

The emissions testing was performed from July 18 to August 5, 1988. Source testing teams from the ARB and ESA performed parallel and simultaneous testing at the boiler outlet (inlet to the air pollution control equipment) and the stack (see Figure 2). Sampling for criteria and non criteria pollutants was performed. Criteria pollutants measured were found to be in compliance with permit limits. This paper will focus on the emission results of dioxins, furans, metals, and PAHs.

TESTING SCHEDULE

Triplicate runs for dioxins and furans, metals, and PAH's were performed for both types of fuel. The only exception was the PAH testing for the 60/40 residential/commercial mixture where only duplicates were performed due to time constraints since the residential mixture was burned for one week only. Additionally, ARB's and ESA's stack sampling team were always within 5 minutes of each other in collecting samples at the stack. However, due to high grain loadings, difficult sampling port positions, and long probes, simultaneous sampling at the inlet was more difficult. Sampling at the inlet was as much as 105 minutes later than at the stack, but the inlet sampling teams always started within 30 minutes of each other.

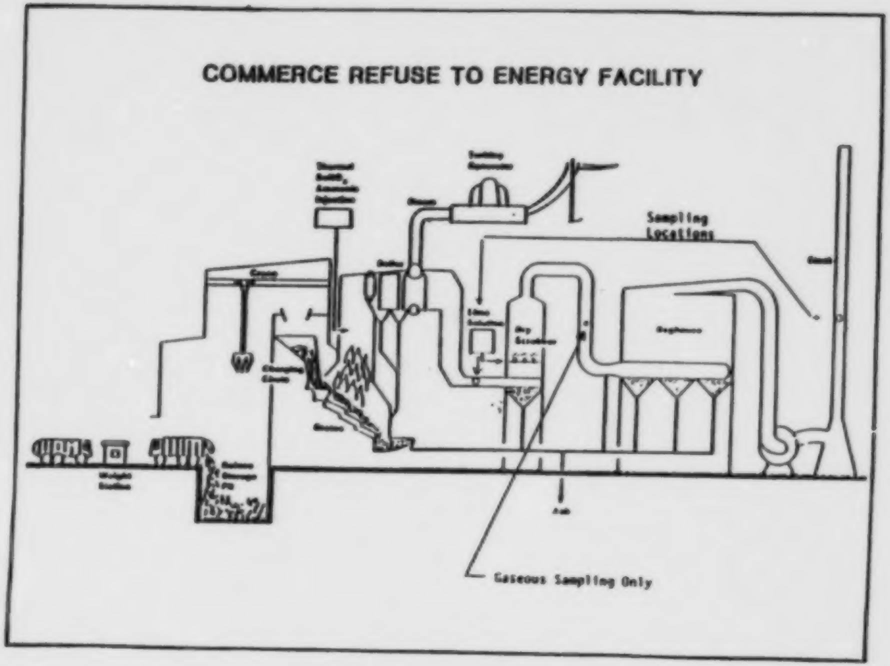
SAMPLING METHODS

Both ESA and ARB followed the sampling procedures outlined below:

Pollutant	Method
Dioxin/furans	EPA Modified Method 5
PAH	EPA Modified Method 5
Metals	EPA Method 5

TABLE 4: Concentrations of Dioxins and Furans in Combined and Fly Ashes

	Residential (avg ppm)			Commercial (avg ppm)		
Congener	combined	baghouse	scrubber	combined	baghouse	scrubber
TCDF	1.5	19.0	2.2	2.61	11.0	3.5
PCDF	1.1	24.0	2.0	1.92	12.0	2.0
HxDCF	1.2	18.0	2.4	2.03	16.0	4.8
HpCDF	0.55	6.0	-	2.14	8.2	2.6
OCDF	-	1.6	-	3.1	4.5	2.0
TCDD	0.12	1.1	-	0.24	0.81	-
PCDD	0.51	2.4	-	0.49	0.6	-
HxCDD	0.76	6.6	-	1.74	3.2	-
HpCDD	0.93	7.3	0.79	2.67	6.0	1.5
OCDD	1.43	5.8	-	4.38	7.0	3.5



COMMERCE REFUSE TO ENERGY FACILITY

Crane
Boiler
Sampling Locations
Dry Scrubber
Baghouse
Stack
Ash
Gaseous Sampling Only

Figure 2. Sampling Locations

DIOXINS, FURANS, AND PAHs

Dioxin, furan, and PAHs were collected using a modified method 5 train. The spent trains were dismantled and samples recovered in the field. ARB's samples were analyzed by ENESCO California Analytical Laboratories and ESA's samples were analyzed by Triangle Laboratories.

METALS

ARB and ESA collected the metals using a method 5 train. The following changes were made to the standard train;

1) The sampling probes were glass or glass lined to avoid probe metal contamination;
2) The glass fiber filter was teflon coated to avoid interference from the filter material;
3) The first two impingers were filled with nitric acid instead of water to collect the volatile metals; and
4) The third impinger was filled with potassium permanganate to collect mercury.

The only exception to this was the chromium sampling. Chromium was also sampled by a dedicated method 5 train containing sodium hydroxide in the first two impingers instead of water. The sodium hydroxide would stabilize and prevent reduction of the hexavalent chromium.

QUALITY ASSURANCE/QUALITY CONTROL (QA/QC)

Both ARB and ESA have established procedures to document the accuracy of the reported sampling and analytical results. These procedures included the use of sampling blanks, field spikes, leak checks, and field and laboratory chain of custody techniques. In addition to the standard QA/QC techniques, ARB and ESA conducted a fourth dioxin and furan test at the stack during the last week of testing. The train was dismantled in the field according to ARB and ESA procedures. ARB and ESA then exchanged sampling trains. ARB and ESA then sent the exchanged trains to their respective laboratories for analysis. This was done to determine the comparability between laboratories.

EMISSIONS TESTING RESULTS

FACILITY OPERATIONS DATA

The facility ran at or near normal conditions for all test days except for two days - 7/18 and 7/28. On 7/18, the plant had switched from burning its normal fuel mixture to the 60% residential/40% commercial fuel mixture. This fuel with its higher moisture content and density required significant operational changes and the facility operator needed most of the day to achieve stable combustion. The unstable combustion is verified by the higher CO emissions and lower steam flowrate. On 7/28, one bag in the baghouse was discovered to have fallen off its support. This would explain the higher metal emissions for this day. Average operations data for the dioxin, metals, and PAHs runs are listed below.

TEST	Furnace O2 (%)	Stack O2 (%)	Steam Flow (1000 lb/hr) (F)	Temp after Econo. (F)	Temp after Scrubber	NOx at 3%	CO O2
Dioxins	6.2	9.5	110.8	518	301	139	34
7/18	10.2	13.0	65.4	494	304	213	115
Metals	6.1	9.1	114	520	301	119	33
PAHs	6.2	9.4	114	529	300	151	38

Other process data collected showed that during the testing the ammonia flowrate operated between 17 to 30 lbs/hr with peaks to 49 lbs/hr. The lime feed rate to the slaker was 360 lbs/hr. The agglomerate feed rate to the dry venturi was fixed at 75 lbs/hr. The pressure drop in the baghouse averaged 5.2 inches of water, and ranged between 3.6 and 7.9 inches over the testing period.

EMISSIONS RESULTS

Table 5 summarizes the ARB and ESA emissions testing results for dioxin, furans, PAH's, and metals. The results in the table reflect the assumptions that compounds not detected in a particular sample were present at one half the detection limit. These values were used to determine average and total emissions.

Both ARB and ESA detected concentrations of dioxin, furans, PAHs and metals similar to that measured at other waste to energy facilities. The ESA results are generally lower in concentrations than the ARB results. Additionally, it appears that the difference in results between ARB and ESA is generally larger than the difference in emissions between the two fuel mixtures.

DIOXIN AND FURAN

Normalizing the dioxin and furan emission results to a California Department of Health Services toxic equivalent basis shows that the differences between fuels and testing groups less than an order of magnitude, and air pollution control equipment efficiencies greater than 99%. Results from testing at the stack with the facility burning the 60% residential, 40% commercial fuel mix were very similar between the two testing groups with ARB results about twice that of ESA. A greater variability between the two groups is evident for the mostly commercial fuel mixture with ARB's results about 6 times greater than ESA's.

TABLE 5
SUMMARY OF EMISSIONS TESTING

AVERAGE PCDD/PCDF CONCENTRATIONS
(ng/Nm3 at 12% CO2)

	60/40 MIX *				95/5 MIX			
	INLET		STACK		INLET		STACK	
	ARB	ESA	ARB	ESA	ARB	ESA	ARB	ESA
TOTAL PCDD/F	414	577**	2.57	1.39	4255	810	30.8	3.32
TOXIC EQUIV.								
CA DHS	37.9	42.1	0.28	0.14	112	53.9	0.94	0.16
EADONS	16.3	17.8	0.11	0.05	46.7	20.7	0.34	0.07
EPA	7.54	9.61	0.06	0.02	24.6	10.5	0.21	0.04

* - average of 7/20 and 7/22 tests since 7/18 test not at normal operation.
**- results of 7/22 test only due to interferences in 7/20 test analysis.

PCDD/PCDF TEST ON 7/18
(ng/Nm3 @ 12% CO2)

	INLET		STACK	
	ARB	ESA	ARB	ESA
TOTAL PCDD/F	1192	253.8	182.1	27.8
TOXIC EQUIV.				
CA DHS	55.0	16.6	13.5	0.68
EADONS	23.3	6.06	6.01	0.25
EPA	12.3	2.92	3.50	0.17

TABLE 5
SUMMARY OF EMISSIONS TESTING
(continued)

PAH
(ug/Nm3 @ 12% CO2)

	60/40 MIX				95/5 MIX			
	INLET		STACK		INLET		STACK	
	ARB	ESA	ARB	ESA	ARB	ESA	ARB	ESA
TOTAL	18.6	31.4	7.49	0.46	19.6	7.46	3.08	1.30
TOTAL less naphthalene	17.5	25.4	0.071	0.14	9.54	3.80	0.58	0.088

METALS
(ug/Nm3 @ 12% CO2)

	60/40 MIX				95/5 MIX			
	INLET		STACK		INLET		STACK	
	ARB	ESA	ARB	ESA	ARB	ESA	ARB	ESA
As	159.4	78.0	0.22	0.16	132.9	68.9	0.28	0.10
Be	N/A	6.88	N/A	0.12	N/A	3.81	N/A	0.09
Cd	1860	1677	0.29	1.97	921.3	930	0.02	0.40
Cr	383.2	3616	1.14	2.33	257	730	0.24	0.45
Pb	16273	18132	2.87	1.97	13216	17949	0.13	4.00
Ni	754.2	4231	10.6	6.32	140	2307	0.36	0.24
Hg	57.9	475.0	1.66	31.13	2.07	276.0	4.49	71.2
Cr (Cr train)	N/A	N/A	N/A	N/A	N/A	931	14.3	0.24
Cr(VI)	N/A	N/A	N/A	N/A	N/A	20.8	0.02	0.21

N/A - not available

TABLE 5
SUMMARY OF EMISSIONS TESTING
(continued)

7/28 TEST (lost bag)
(ng/Nm3 @ 12% CO2)

	INLET		STACK	
	ARB	ESA	ARB	ESA
As	59.8	92.0	0.29	1.38
Cd	765.1	1330	22.4	18.0
Cr	154.3	1027	16.8	3.63
Pb	17281	20400	318.4	366.0
Ni	283.9	840	86.9	2.60
Hg	2.22	332	1.34	85

Of particular interest are the results of the dioxin testing performed on the first day where the facility was running at less than optimal combustion. The toxic equivalent emissions for this test compared to the average of the other test burning the 60/40 mixture of fuel is 50 and 5 times greater for ARB and ESA results respectively. The air pollution control equipment efficiency dropped to 75% and 96% using ARB and ESA results respectively.

PAHs

Both ESA and ARB had high naphthalene concentrations in their blank trains and Triangle Labs informed ESA that naphthalene is a decomposition product of the adsorbent used in the PAH sampling trains. Therefore, it appears more appropriate to compare the PAH results as the total less the naphthalene concentration.

The total PAH less naphthalene emissions for the 60/40 mixture are slightly higher for both testing teams. The efficiency for the air pollution control equipment was better than 90% for both fuels. The differences in the results between the two testing teams are greater than the differences each testing team found between fuels. ARB results are about 5 to 6 times higher than those measured by ESA.

METALS

The metal emission results show that most metals are controlled at a better than 99% efficiency. The inlet and stack concentrations of lead and zinc are higher for the mostly residential fuel mixture, with the exception of ESA's lead stack concentration which is two times less than the mostly commercial fuel result. This correlates well with the waste and ash results which show a higher zinc and lead concentration for the mostly residential fuel mixture. Other metals that appear to have higher emissions rates from mostly residential fuel are cadmium, chromium, and nickel. Additionally, there are large differences in emissions between the testing teams. For example, ESA's average lead emission at the stack while the facility was burning mostly commercial fuel was 31 times higher than ARB's.

It is interesting to note the change in emissions of metals when one bag in the baghouse fell from its supports which occurred during the 7/28 metals test. Several of the metals showed a 10 to 100 fold increase in emission rate.

QA/QC

The results of the dioxin train exchange performed as an additional QA/QC measure show that ESA's train analyzed by ENESCO Labs has a California DHS toxic equivalent value approximately 2 times higher than ARB's sample which was analyzed by Triangle Labs.

CONCLUSION

The completion of the testing at the Commerce facility has provided current data on a state-of-the-art facility, which can be used not only in evaluating the Commerce facility, but also in evaluating proposed facilities. It has also provided some comparisons between fuels, ash, testing teams, and laboratories. However, it is still early in the data analysis and more review of the data need to be completed. It is hoped that correlations between the three aspects of the testing program (fuel, ash and emissions) can be determined.

Preliminary analysis indicates that, 1) emissions of some metals correlate well between the fuel, ash, and emissions, 2) significant amounts of soluble metals exist in the ash, and 3) the difference between the two fuel mixtures are not great. Preliminary analysis of the air emissions results indicate, 1) the differences in emissions measured are less due to fuel mix, and more due to testing team and laboratory, 2) the results of the two teams should be viewed as valid and representative of the emissions associated with this facility since no technical reason exists to explain the difference, and 3) the need for careful facility maintenance and operation in avoiding higher concentrations of non-criteria pollutant emissions.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

Evaluation of the Potential Impact of the Detroit Incinerator on Air Quality in Windsor Ontario

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ABSTRACT

In 1986, the Greater Detroit Resource Recovery Authority announced that a municipal solid waste (MSW) incinerator, the largest in the world, would be constructed in Detroit, approximately 6 kilometres from the Canada - U.S. border and the City of Windsor, Ontario. The incinerator would be equipped with electrostatic precipitators for air pollution control, rather than a wet scrubber/fabric filter combination considered to be Best Available Control Technology. Windsor residents became concerned over the impact of the incinerator emissions on their air quality.

Environment Canada, in co-operation with the Ontario Ministry of Environment and the Walpole Island Indian Band, has established an ambient air toxics monitoring network to measure air pollutants associated with MSW incineration, including dioxins, furans, PAH, PCB, CP, CB and PM10.

The network has been in operation since July 1988, collecting background data. These data, and the data collected subsequent to the incinerator start-up in May 1989, will be evaluated in conjunction with inventory and source characterization information, and assessed to determine the impact on Windsor ambient air quality.

INTRODUCTION

The City of Windsor, with a population of over 250,000, is located in the southwest corner of the province of Ontario. Covering an area of approximately 30,000 acres, Windsor is the fifth-largest manufacturing centre in Canada, with such major industries as motor vehicle and related parts manufacturing, and tool and die making. With a climate comparable to that of Northern California, the Windsor-Essex County area enjoys the longest growing season in Canada (1). To the west and north of Windsor, across the Detroit River, lies the City of Detroit.

In 1986, the Greater Detroit Resource Recovery Authority (GDRRA) announced that a municipal waste incinerator would be constructed in downtown Detroit, approximately 6 kilometres (4 miles) from the Canada-U.S. border. This facility, one of the largest in the world, would process up to approximately 3,300 tons per day of raw municipal waste into a refuse-derived fuel, which would then be burned to produce steam for heating and electricity. The scheduled start-up date for the Detroit Incinerator is May 1989. The incinerator would be equipped with three five-field electrostatic precipitators for control of particulate air pollutant emissions. However, both Environment Canada and the United States Environmental Protection Agency have identified Best Available Control Technology (BACT) for municipal waste incinerators as a combination of dry scrubber, for acid and other gaseous emission control, and a fabric filter, for particulate emission control (2)(3).

Faced with the prospect of a major waste incineration facility without BACT immediately upwind of their city, Windsor residents became concerned that the quality of their air would be adversely affected. This concern was expressed to the Canadian Government, which on a number of occasions raised objections to the lack of BACT on the GDRRA Incinerator. Environment Canada's involvement came about as a result of a suggestion that Canadian arguments in favour of BACT would be better supported with some evidence of adverse impact. Environment Canada's Conservation and Protection Ontario Region office proceeded to establish the Windsor/Walpole Ambient Air Toxics Monitoring Network. The network would measure the ambient air concentrations of toxic organic and other compounds associated with the combustion of municipal solid waste (MSW), both before and after the start-up of the Detroit Incinerator, in order to evaluate the impact of the incinerator on air quality in and around Windsor.

DESCRIPTION OF THE WINDSOR/WALPOLE AMBIENT AIR MONITORING NETWORK

STATION LOCATIONS

The Windsor/Walpole Ambient Air Monitoring Network consists of two stations, one located in downtown Windsor and the other located on the Walpole Island Indian Reserve, approximately 50 kilometres (30 miles) directly northeast of Windsor. The Windsor station is located on the roof of the Federal Government Building at 471 University Avenue, the site of an existing National Air Pollution Surveillance Network (NAPS) station operated jointly by Environment Canada and the Ministry of Environment for the province of Ontario (MOE). This site was chosen because of its location in the area of Windsor that is predicted by the MOE modelling studies to be the area most impacted by emissions from the Detroit Incinerator. The Windsor station is operated and maintained by MOE technicians, and has been in operation since July 1987 (4).

The Walpole station is located on St. Anne's Island within the Walpole Island Reserve, and is a stand-alone station, constructed exclusively for the Windsor/Walpole network. The station is operated by the members of the Walpole Band who staff the Walpole Island Environmental Research Centre. They have been specially trained by Environment Canada in the operation and maintenance of the sampling equipment. The Walpole site was chosen to represent background conditions, as it is a rural setting without nearby industrial or urban influences. The Walpole station has been in operation since January 1988 (5).

Figure 1 shows the locations of the Windsor and Walpole stations, as well as the location of the Detroit Incinerator.

TARGET PARAMETERS

The stations are equipped with specialized instruments for monitoring such air toxics as polycyclic aromatic hydrocarbons (PAH), polychlorinated biphenyls (PCB), chlorinated phenols (CP) and chlorinated benzenes (CB), polychlorinated dibenzo-p-dioxins (PCDD), polychlorinated dibenzo-furans (PCDF), volatile organic compounds (VOC) and particulate matter (PM10). Most of these compounds are known to be components of the MSW incinerator stack emissions, to one degree or another (3)(4). Table 1 provides a complete listing of the individual parameters of interest to the network. Because they are not a major parameter, and because the existing emission database is small, VOC are not discussed in this paper. The Windsor/Walpole monitoring effort is the most comprehensive air toxics sampling program currently underway in Canada. For many of the compounds measured, it is the first time in Canada that an attempt has been made to identify and quantify them in ambient air (5).

Since their start-up, both stations have been measuring background levels of the target parameters in the ambient air.

TABLE 1. WINDSOR/WALPOLE AIR MONITORING NETWORK LIST OF PARAMETERS

POLYCYCLIC AROMATIC HYDROCARBONS	INHALABLE PARTICULATE (PM10)	
Acenaphthylene	Aluminum	Selenium
Acenaphthene	Silicon	Bromine
Fluorene	Phosphorus	Rubidium
2-Methylfluorene	Sulphur	Strontium
Phenanthrene	Chlorine	Yttrium
Anthracene	Potassium	Zirconium
Fluoranthene	Calcium	Molybdenum
Pyrene	Titanium	Palladium
Benzo(a)Fluorene	Vanadium	Silver
Benzo(b)Fluorene	Chromium	Cadmium
3-Methylpyrene	Lead	Indium
Benzo(g,h,i)Fluoranthene	Manganese	Tin
Benzo(a)Anthracene	Iron	Antimony
Chrysene and Triphenylene	Nickel	Barium
7-Methylbenzo(a)Anthracene	Copper	Sulphate
Benzo(b&k)Fluoranthene	Zinc	Nitrate
Benzo(e)Pyrene	Gallium	Nitrite
Benzo(a)Pyrene	Arsenic	Phosphate
Perylene		
3-Methylcholanthrene		
Indeno(1,2,3-c,d)Pyrene	PCDD (Dioxins) PCDF (Furans)	
Dibenzo(a,c)&(a,h)Anthracene		
Benzo(b)Chrysene	CHLORINATED BENZENES (CB) CHLORINATED PHENOLS (CP) PCB	
Benzo(g,h,i)Perylene		
Anthanthrene		

SAMPLING METHODOLOGY

Table 2 shows the protocols used for sampling and analysis of each compound group. PCDD, PCDF, PAH, PCB, CB, and CP samples are collected using a custom modified Hi-Vol sampling system, equipped with a Teflon-coated glass fibre filter for the particulate phase and either a PUF

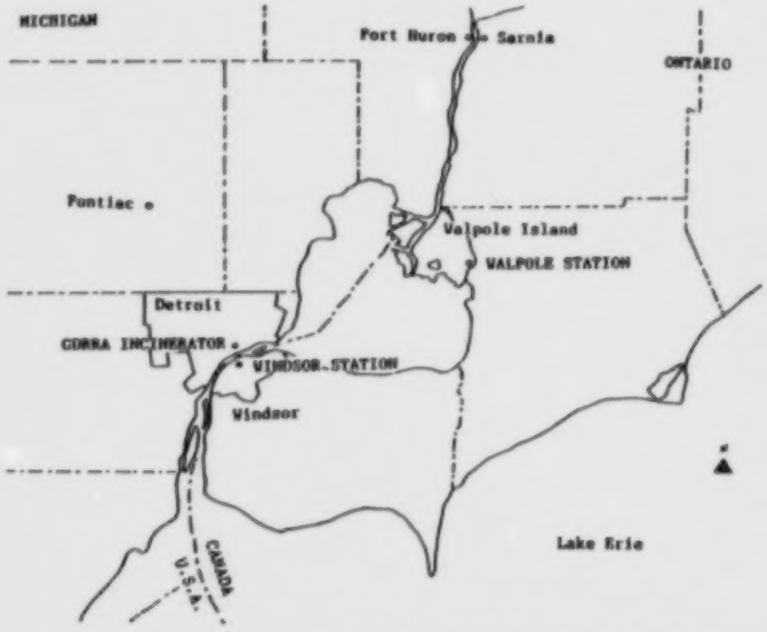

MICHIGAN
Port Huron
Sarnia
ONTARIO
Pontiac
Walpole Island
WALPOLE STATION
Detroit
CORRA INCINERATOR
WINDSOR STATION
Windsor
CANADA
U.S.A.
Lake Erie

Figure 1. Windsor/Walpole Air Monitoring Network Locations Of Monitoring Stations

(polyurethane foam) adsorbent trap or a PUF/XAD-2/PUF resin "sandwich" adsorbent trap for the vapour phase. Figure 2 illustrates a typical Hi-Vol sampler assembly. Figure 3 depicts a PUF canister assembly. The PUF or PUF/XAD-2/PUF sorbent, contained within a glass tube, is inserted into the canister. PM10 samples are collected using a Sierra Andersen dichotomous sampler, with a Ghia Corporation Teflon membrane filter (2 um pore size), as shown in Figure 4. The VOC samples are collected using Summa-polished evacuated canisters. Both stations are equipped with two modified Hi-Vols, one evacuated canister and one dichotomous sampler. The Walpole station is also equipped with a meteorological station. The Windsor station obtains its meteorological data through the MOE air monitoring network.

TABLE 2. WINDSOR/WALPOLE AIR MONITORING NETWORK SAMPLING/ANALYTICAL PROTOCOLS

Compound	No. of Species or Isomer Groups Measured	Sampling Method	Analytical Method	Sample Volume (24h) m3	Target Detection Level	Sample Collectd
Dioxin/Furan	10	Hi-Vol/PUF	GC-MS	1000	0.05-0.20 pg/m3	every 30 d
PCB	10	Hi-Vol/PUF-XAD	GC-MSD	500	0.04-0.08 ng/m3	every 12 d
Chloro-benzenes (1)	4	Hi-Vol/PUF-XAD	GC-MSD	500	0.02 ng/m3	every 12 d
Chlorophenols	4	Hi-Vol/PUF-XAD	GC-MSD	500	0.06-0.12 ng/m3	every 12 d
PAH	25	Hi-Vol/PUF or PUF/XAD	GC-MSD	1000 or 500	0.02-0.05 ng/m3	every 12 d
Particulates (PM10)	--	Dichotomous	Gravi-metric	24	--	every 6 d
Metals & Ions	42	Dichotomous	EDXRF&IC	24	1 - 3 ng/m3	every 6 d
VOC	50	Evacuated Canister	GC-FID/MSD	16 L	0.1 ug/m3	every 6 d

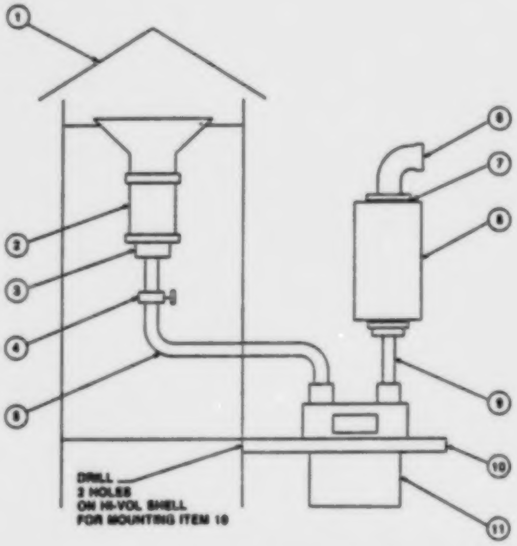
(1) Di-chlorobenzenes sampled as VOC

NOTE: PCB, CP, CB sampled at the same time with the same sampler. Split extract scheme used. At times, PAH also determined from the same sample (see Table 3).

PCDD/PCDF samples are collected once every 24 days. PAH, PCB, CP and CB samples are collected every 12 days. VOC and PM10 samplers operate every sixth day. Samples are collected over a 24-hour period, from midnight to midnight. Both stations follow the same schedule. This schedule, shown in part in Table 3, is also arranged to correspond to a similar sampling program in Detroit, conducted for the GDRRA by Roy F. Weston, Inc. (6). QA/QC procedures include field blank samples, which are run on the modified Hi-Vols once every 30 days for 24 hours. Samples are transported to Environment Canada's River Road Environmental Technology Centre in Ottawa for preparation and analysis.

ANALYTICAL METHODOLOGY

All samples are extracted, concentrated (where necessary), cleaned and


①
⑥
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⑧
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④
⑨
⑤
⑩
DRILL
2 HOLES
ON HI-VOL SHELL
FOR MOUNTING ITEM 10
⑪

① HI-VOL SAMPLER SHELL WITH CONTROL TIMER AND PRESSURE SENSOR

② PUF CANISTER ASSEMBLY DWG NO. 1801

③ CANISTER ADAPTOR DWG NO. 2001

④ CONNECTOR DWG NO. 2010

⑤ S-PIPE ASSEMBLY

⑥ PIPE ELBOW AND FLANGE DWG NO. 2030

⑦ RUBBER GASKET DWG NO. 2040

⑧ HI-VOL MOTOR HOUSING DWG NO. 2050

⑨ MOTOR ADAPTOR DWG NO. 2060

⑩ GAS METER SUPPORT BRACKET DWG NO. 2070

⑪ GAS METER ROCKWELL MODEL RCM 415

Figure 2. Windsor/Walpole Air Monitoring Network Modified Hi-Vol Sampler Assembly

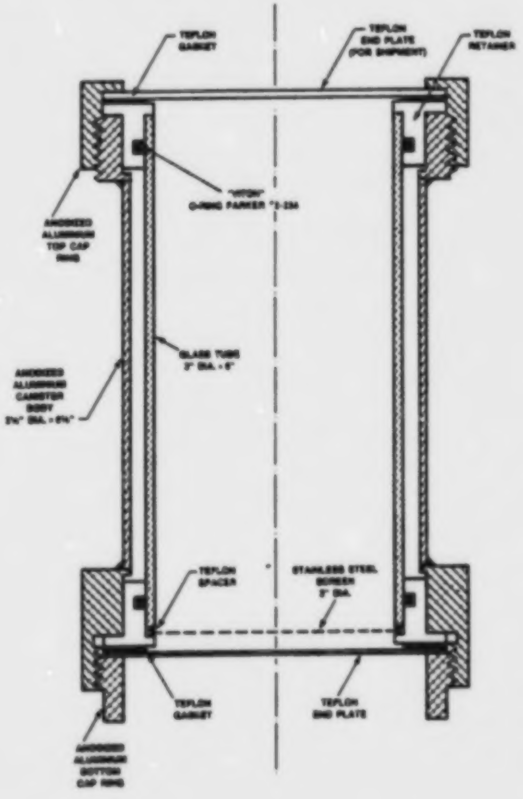
TEFLON GASKET
TEFLON END PLATE (FOR SHIPMENT)
TEFLON RETAINER
"VITON" O-RING PARKER "2-234
ANODIZED ALUMINUM TOP CAP RING
GLASS TUBE 2" DIA. × 5"
ANODIZED ALUMINUM CANISTER BODY 2¼" DIA. × 5¾"
TEFLON SPACER
STAINLESS STEEL SCREEN 2" DIA.
TEFLON GASKET
TEFLON END PLATE
ANODIZED ALUMINUM BOTTOM CAP RING

Figure 3. Windsor/Walpole Air Monitoring Network PUF/PUF-XAD Canister Assembly

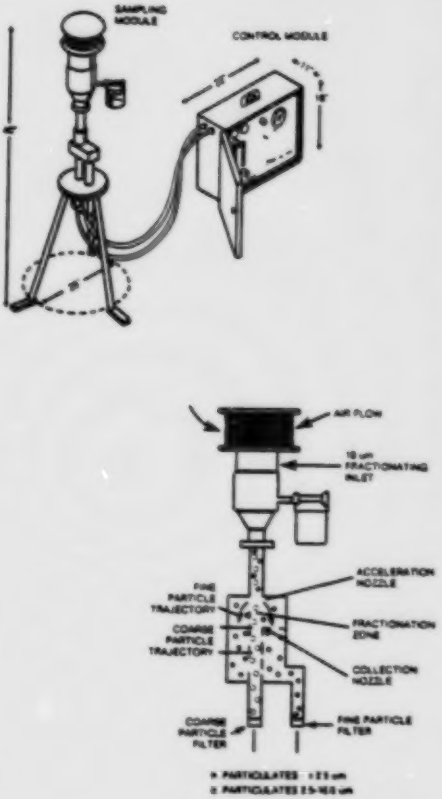
SAMPLING MODULE
CONTROL MODULE
AIR FLOW
10 um FRACTIONATING INLET
ACCELERATION NOZZLE
FINE PARTICLE TRAJECTORY
FRACTIONATION ZONE
COARSE PARTICLE TRAJECTORY
COLLECTION NOZZLE
COARSE PARTICLE FILTER
FINE PARTICLE FILTER
• PARTICULATES <2.5 um
○ PARTICULATES 2.5-10.0 um

Figure 4. Windsor/Walpole Air Monitoring Network Dichotomous Sampler

then analyzed. Particulate and vapour fractions are extracted separately, for most samples. PCDD/DF samples are analyzed using a Finnigan 4500 HR GC/LRMS gas chromatograph-mass spectrometer system (GC/MS). PAH, PCB, CB and CP samples are analyzed using a Hewlett-Packard 5890/5970 GC/MSD gas chromatograph-mass selective detector system (GC/MSD). PM10 samples are analyzed gravimetrically. Metals and ions concentrations are determined using energy dispersive X-ray fluorescence and ion chromatography (EDXRF&IC). VOC samples are concentrated and analyzed using a Hewlett-Packard 5890 gas chromatograph with flame ionization detector, and by mass selective detection (GC-FID/MSD). Table 2 summarizes the analytical methodologies used for each compound group, and the corresponding target detection levels (4).

TABLE 3. WINDSOR/WALPOLE AIR MONITORING NETWORK
TYPICAL SAMPLING SCHEDULE

Sample Day #	Date	Semi-Volatile Organics		Volatile Organics Sampler	Dichotomous Sampler
		Sampler #1	Sampler #2		
1	Jan. 4/89	---	PAH/PCB/CP/CB	VOC	Metals
2	Jan. 10/89	PAH	Field Blank	VOC	Metals
3	Jan. 16/89	PAH/PCB/CP/CB	PCDD/PCDF	VOC	Metals
4	Jan. 22/89	PAH	---	VOC	Metals
5	Jan. 28/89	---	PAH/PCB/CP/CB	VOC	Metals
6	Feb. 3/89	PAH	---	VOC	Metals
7	Feb. 9/89	PAH/PCB/CP/CB	PCDD/PCDF	VOC	Metals
8	Feb. 15/89	PAH	---	VOC	Metals
9	Feb. 21/89	---	PAH/PCB/CP/CB	VOC	Metals
10	Feb. 27/89	PAH	Field Blank	VOC	Metals

INTERNATIONAL INVOLVEMENT

As mentioned previously, the Windsor/Walpole network operating schedule corresponds to the operating schedule of the GDRRA monitoring stations. The GDRRA has established two ambient air monitoring stations in the Detroit area to monitor the local impact of the Detroit Incinerator (6). Environment Canada, along with the MOE and the City of Windsor, is currently involved in a technology transfer arrangement with Wayne County, the GDRRA, the Michigan Department of Natural Resources and the U.S. EPA concerning the Detroit Incinerator. This arrangement, called the Ad Hoc Committee for Ambient Air Monitoring for the Detroit Incinerator, met first in March 1987, and has since offered an excellent forum for the exchange of information on ambient air toxics sampling and analytical methodology. The Ad Hoc Committee also provides the opportunity to discuss the monitoring results of the participating networks. It is a prime working example of international co-operation.

RESULTS

Following is a summary of the results from the first year of operation of the Windsor/Walpole network, by compound group. Due to operational problems, samplers did not always run on scheduled sampling days. The results presented below are those obtained from the available samples.

Sample results that fall below compound detection levels are reported in the tables as Not Detected (ND). Where individual sample results are found to be below the detection level (ND) for a particular compound, the mean concentration for that compound is calculated using a value of one-half the detection level for each of the ND results. Where the detection level is a range, the mean of the range is used as the detection level in the calculation. If the calculated mean compound concentration falls below detection levels, or if greater than 50% of the data are below detection level, the mean is not reported.

Standard deviations are calculated based on ND values in the data base set to zero (ND=0). Standard deviations are reported only if greater than 50% of the data are above detection level. Compound group totals are also calculated based on ND=0, and are not necessarily equal to the sum of the means of the individual compounds in each group.

All concentrations reported are totals of vapour and particulate phase concentrations, with the exception of the PM10 results, the totals of which are the sums of the fine and coarse fractions collected.

POLYCHLORINATED DIBENZO-P-DIOXINS (PCDD) AND POLYCHLORINATED DIBENZOFURANS (PCDF)

PCDD and PCDF results are available for 13 sampling days for the Windsor station, and 8 sampling days for the Walpole station. Results are reported by homologue subgroup. Five homologue groups, based on degree of chlorination (Cl4 through Cl8) are reported for PCDD and PCDF. Summary statistics for each dioxin and furan homologue group and for total dioxins and furans are contained in Table 4 for Windsor and in Table 5 for Walpole.

TABLE 4. WINDSOR/WALPOLE AIR MONITORING NETWORK
MEASURED CONCENTRATIONS OF POLYCHLORINATED DIBENZO-P-DIOXINS
AND POLYCHLORINATED DIBENZOFURANS - WINDSOR
(pg/m3)
13 SAMPLE DAYS

Compound	No> Detection	Mean Detection Level	Mean	Median	Maximum
Tetra-CDD	1	0.06	-	ND	0.03
Penta-CDD	2	0.16	-	ND	0.15
Hexa-CDD	8	0.16	0.31	0.23	1.66
Hepta-CDD	11	0.21	0.63	0.42	2.15
Octa-CDD	13	0.20	1.19	0.90	3.29
Total PCDD			2.12	1.42	7.09
Tetra-CDF	9	0.11	0.21	0.19	0.78
Penta-CDF	6	0.08	0.09	ND	0.31
Hexa-CDF	6	0.10	0.10	ND	0.30
Hepta-CDF	5	0.09	0.08	ND	0.19
Octa-CDF	7	0.20	0.13	0.12	0.36
Total PCDF			0.46	0.40	1.19
ND = less than Mean Detection Level as given in Column 3					

Mean total PCDD concentrations were 2.12 pg/m3 at Windsor and 0.51 pg/m3 at Walpole. Hepta- and octa-dioxins accounted for over 85% of the total measured dioxins at both stations. Tetra- and penta-dioxins were not found above their respective detection levels at either Windsor or Walpole. The Walpole station also did not detect hexa-dioxin.

The mean total PCDF concentration was 0.46 pg/m3 at Windsor. Tetra-, penta- and hexa-furan accounted for a majority of the total measured mean furan concentrations, with tetra-furan having the highest mean and maximum concentration values. With the exception of one tetra-furan sample result, no furans were found at greater than detection level at Walpole.

TABLE 5. WINDSOR/WALPOLE AIR MONITORING NETWORK
MEASURED CONCENTRATIONS OF POLYCHLORINATED DIBENZO-P-DIOXINS
AND POLYCHLORINATED DIBENZOFURANS - WALPOLE
(pg/m3)
8 SAMPLE DAYS

Compound	No> Detection	Mean Detection Level	Mean	Median	Maximum
Tetra-CDD	0	0.07	-	ND	ND
Penta-CDD	0	0.19	-	ND	ND
Hexa-CDD	1	0.18	-	ND	0.10
Hepta-CDD	4	0.17	0.19	ND	0.40
Octa-CDD	6	0.23	0.38	0.29	0.96
Total PCDD			0.68	0.29	1.30
Tetra-CDF	1	0.05	-	ND	0.12
Penta-CDF	0	0.07	-	ND	ND
Hexa-CDF	0	0.10	-	ND	ND
Hepta-CDF	0	0.07	-	ND	ND
Octa-CDF	0	0.14	-	ND	ND
Total PCDF			-	ND	0.12

ND = less than Mean Detection Level as given in Column 3

Concentrations of both dioxins and furans at the Walpole station were 4 to 20 times less than at the Windsor station.

There are no national air quality objectives for PCDD or PCDF in Canada. The province of Ontario has a provisional ambient air guideline for 2,3,7,8-TCDD of 30 pg/m3 (not to be exceeded on average over a year) (7). None of the measured concentrations at either station exceed this guideline. No other measurements of ambient air PCDD/DF have been reported for Canadian urban or rural areas. A 1985/86 sampling program in West Germany at 18 sites showed a mean concentration for total PCDD of 3.2 pg/m3 and for total PCDF of 5.5 pg/m3 (8).

POLYCYCLIC AROMATIC HYDROCARBONS (PAH)

PAH results are available for 55 sampling days for the Windsor station and 26 sampling days for the Walpole station. Tables 6 and 7 present the summary statistics for each of the 28 measured compounds for Windsor and Walpole, respectively.

Phenanthrene and fluoranthene had the highest mean concentrations at both stations. Acenaphthylene, acenaphthene, fluorene, 2-methyl fluorene anthracene and pyrene were also found in high concentrations, relative to other PAH, at both stations. For the higher molecular weight compounds, benzo(b)fluoranthene and chrysene (+ triphenylene) were found in the highest concentrations at both stations. In general, PAH concentrations measured at the Walpole station were approximately one order of magnitude less than those measured at the Windsor station.

To date, national ambient air quality standards have not been established for benzo(a)pyrene or other polycyclics. The State of Michigan has established an annual average environmental acceptability level for PAH (as benzo(a)pyrene) of 0.3 ng/m3 (9). The environmental acceptability level is considered a guideline and is used in screening permit applications for

TABLE 6. MEASURED CONCENTRATIONS OF POLYCYCLIC AROMATIC HYDROCARBONS -WINDSOR (ng/m3) 55 SAMPLING DAYS

Compound	No> Detection	Mean	Standard Deviation	Median	Maximum
Acenaphthylene	55	5.12	6.22	3.04	27.15
Acenaphthene	54	6.28	11.25	2.29	59.99
Fluorene	55	6.83	6.71	3.72	30.74
2-Me-Fluorene	54	5.18	2.71	4.53	13.84
Phenanthrene	55	33.51	24.90	26.49	136.97
Anthracene	38	3.28	5.30	0.83	24.83
Fluoranthene	55	10.55	8.01	7.75	39.02
Pyrene	55	6.58	4.83	4.76	22.00
Benzo(a)Fluorene	54	1.14	1.19	0.75	5.91
Benzo(b)Fluorene	51	0.70	0.94	0.31	4.04
1-Me-Pyrene	49	0.30	0.25	0.21	0.99
Benzo(g,h,i)Fluoranth	55	0.65	0.52	0.45	2.38
Benzo(a)Anthracene	53	0.88	1.19	0.35	5.33
Chrysene & Triphenyle	55	1.87	2.06	0.92	11.18
7-Me-Benzo(a)Anthrace	0	-	-	ND	ND
Benzo(b)Fluoranthene	55	2.92	2.56	2.08	10.72
Benzo(k)Fluoranthene	2	-	-	ND	1.85
Benzo(e)Pyrene	55	0.78	0.73	0.51	3.08
Benzo(a)Pyrene	55	0.61	0.65	0.37	3.32
Perylene	40	0.09	0.12	0.06	0.69
2-Me-Cholanthrene	3	-	-	ND	0.05
Indeno(1,2,3-c,d)Pyren	53	1.04	1.05	0.61	5.23
Dibenzo(a,c)&(a,h)A	19	-	-	ND	0.57
Benzo(b)Chrysene	20	-	-	ND	0.38
Benzo(g,h,i)Perylene	54	0.75	0.63	0.54	3.00
Anthanthrene	26	-	-	ND	0.26
Total PAH		95.75	80.96	65.60	411.74

ND - less than the average Detection Level of 0.01 - 0.05 ng/m3 for PAH

TABLE 7. MEASURED CONCENTRATIONS OF POLYCYCLIC AROMATIC HYDROCARBONS -WALPOLE (ng/m3) 26 SAMPLING DAYS

Compound	No> Detection	Mean	Standard Deviation	Median	Maximum
Acenaphthylene	24	0.25	0.78	0.06	4.01
Acenaphthene	23	0.37	0.41	0.17	1.72
Fluorene	26	0.85	0.83	0.48	4.01
2-Me-Fluorene	25	0.85	0.50	0.79	1.90
Phenanthrene	26	3.34	2.31	2.77	10.21
Anthracene	3	-	-	ND	2.01
Fluoranthene	26	0.88	0.73	0.70	3.62
Pyrene	26	0.46	0.52	0.35	2.83
Benzo(a)Fluorene	22	0.07	0.10	0.05	0.53
Benzo(b)Fluorene	12	0.04	0.06	ND	0.28
1-Me-Pyrene	7	-	-	ND	0.19
Benzo(g,h,i)Fluoranth	17	0.06	0.10	0.04	0.48
Benzo(a)Anthracene	17	0.05	0.06	0.03	0.26
Chrysene & Triphenyle	23	0.18	0.18	0.15	0.72
7-Me-Benzo(a)Anthrace	0	-	-	ND	ND
Benzo(b)Fluoranthene	23	0.32	0.29	0.27	1.27
Benzo(k)Fluoranthene	0	-	-	ND	ND
Benzo(e)Pyrene	22	0.09	0.07	0.06	0.31
Benzo(a)Pyrene	18	0.07	0.06	0.06	0.19
Perylene	1	-	-	ND	0.04
2-Me-Cholanthrene	0	-	-	ND	ND
Indeno(1,2,3-c,d)Pyren	15	0.07	0.09	0.03	0.40
Dibenzo(a,c)&(a,h)A	3	-	-	ND	0.03
Benzo(b)Chrysene	0	-	-	ND	ND
Benzo(g,h,i)Perylene	17	0.06	0.07	0.06	0.28
Anthanthrene	0	-	-	ND	ND
Total PAH		8.06	6.37	6.74	31.24

ND = less than the average Detection Level of 0.01 - 0.05 ng/m3 for PAH new facilities.

A number of measurement programs for PAH have been carried out in Canadian cities and in other areas of North America. A comparison of some recently reported benzo(a)pyrene concentration data is provided in Table 8. The Walpole station mean concentration is included as a comparison of rural to urban PAH concentrations, and is again approximately one order of magnitude less than the other reported values.

TABLE 8. COMPARISON OF BENZO(A)PYRENE CONCENTRATIONS IN CANADA AND U.S. URBAN AREAS
(ng/m3)

City	No. of Samples	Sampling Period	Mean Concentration (ng/m3)
Windsor	55	July 87 - Sept 88	0.61
Walpole	26	Jan 88 - Sept 88	0.06
Toronto	42	Oct 84 - July 86	0.30
Montreal	29	Oct 84 - May 86	0.60
Vancouver	14	Feb 85 - Mar 85	0.30
Newark, N.J.	30	Summer 1982	0.20 (1)
Columbia, S.C.	13	Feb 81 - April 83	0.50
Los Angeles, CA	~1000	Aug 81 - Mar 82	0.60

(1) Geometric Mean

POLYCHLORINATED BIPHENYLS (PCB), CHLORINATED BENZENES (CB) AND CHLORINATED PHENOLS (CP)

Data for PCB, CB and CP are available for 19 sampling days for the Windsor station, and for 13 sampling days for the Walpole station. Results are reported by degree of chlorination for 10 PCB isomer groups (Cl1 to Cl10), for 4 CB isomer groups (Cl3 to Cl6) and for 4 CP isomer groups (Cl2 to Cl5). Summary statistics for these compounds are provided in Table 9 for the Windsor station and in Table 10 for the Walpole station.

PCB concentrations were low, with a mean total PCB concentration of 0.8 ng/m3 for Windsor. Trichlorobiphenyls and tetrachlorobiphenyls were found in the highest concentrations. Isomer groups Cl1 and Cl8 through Cl10 were not detected. At the Walpole station, trichlorobiphenyls were the only isomers detected, with a maximum concentration of 0.07 ng/m3. This is surprising, as it was expected that, because of the nature of the compound, detectable levels of PCB would be found in ambient air even in a rural setting.

Very few measurements of PCB by isomer group in ambient air have been reported in North America. Total PCB concentrations have been reported to be 1.5 ng/m3 in Columbia, South Carolina, 0.5 ng/m3 in Denver, Colorado and 0.07 ng/m3 in Stockholm, Sweden (10). There is currently no national air quality objective for PCB in Canada. The Ontario annual ambient air quality guideline for PCB is 35 ng/m3 (7). Michigan's annual average environmental acceptability level for PCB (Aroclor 1260) is 1 ng/m3 (9).

Mean total CB concentrations were 3.28 ng/m3 for the Windsor station and 1.9 ng/m3 for the Walpole station. Trichlorobenzene accounted for over 75% of the total at both stations. Hexachlorobenzene is the most toxic compound in the chlorinated benzene class. Mean concentrations of hexachlorobenzene were 0.15 ng/m3 at Windsor and 0.16 ng/m3 at Walpole. Concentrations at the Walpole station were on average one-half those measured at the Windsor station. These concentrations are comparable to total hexachlorobenzene concentrations measured in Columbia, South Carolina (0.29 ng/m3), Denver, Colorado (0.24 ng/m3) and Stockholm, Sweden (0.07 ng/m3) (10). the State of Michigan annual average environmental

TABLE 9. WINDSOR/WALPOLE AIR MONITORING NETWORK MEASURED CONCENTRATIONS OF POLYCHLORINATED BIPHENYLS CHLORINATED BENZENES & CHLORINATED PHENOLS - WINDSOR
(ng/m3)
19 SAMPLING DAYS

Compound	No> Detection	Mean	Standard Deviation	Median	Maximum
Cl1-PCB	0	-	-	ND	ND
Cl2-PCB	10	0.06	0.09	0.03	0.35
Cl3-PCB	19	0.25	0.20	0.16	0.73
Cl4-PCB	16	0.32	0.36	0.19	1.32
Cl5-PCB	16	0.16	0.15	0.13	0.49
Cl6-PCB	7	-	-	ND	0.18
Cl7-PCB	1	-	-	ND	0.06
Cl8-PCB	0	-	-	ND	ND
Cl9-PCB	0	-	-	ND	ND
Cl10-PCB	0	-	-	ND	ND
Total PCB		0.80	0.68	0.63	2.40
Cl3-CB	17	2.64	2.04	2.32	7.53
Cl4-CB	19	0.37	0.55	0.24	2.56
Cl5-CB	19	0.13	0.10	0.10	0.50
Cl6-CB	18	0.15	0.07	0.14	0.27
Total CB		3.28	2.46	2.70	9.72
Cl2-CP	18	0.65	0.59	0.47	2.39
Cl3-CP	13	0.13	0.12	0.12	0.42
Cl4-CP	13	0.25	0.26	0.11	0.82
Cl5-CP	14	0.87	0.92	0.75	3.19
Total CP		1.88	1.64	1.52	4.67

ND = less than the average Detection Level of 0.03 - 0.06 ng/m3 for PCB
= less than the average Detection Level of 0.03 ng/m3 for CB
= less than the average Detection Level of 0.03 ng/m3 for CP

TABLE 10. WINDSOR/WALPOLE AIR MONITORING NETWORK
MEASURED CONCENTRATIONS OF POLYCHLORINATED BIPHENYLS
CHLORINATED BENZENES & CHLORINATED PHENOLS - WALPOLE
(ng/m3)
13 SAMPLING DAYS

Compound	No> Detection	Mean	Standard Deviation	Median	Maximum
Cl1-PCB	0	-	-	ND	ND
Cl2-PCB	0	-	-	ND	ND
Cl3-PCB	2	-	-	ND	0.07
Cl4-PCB	0	-	-	ND	ND
Cl5-PCB	0	-	-	ND	ND
Cl6-PCB	0	-	-	ND	ND
Cl7-PCB	0	-	-	ND	ND
Cl8-PCB	0	-	-	ND	ND
Cl9-PCB	0	-	-	ND	ND
Cl10-PCB	0	-	-	ND	ND
Total PCB		-	-	ND	0.07
Cl3-CB	13	1.48	0.99	1.52	3.90
Cl4-CB	13	0.19	0.10	0.18	0.42
Cl5-CB	12	0.08	0.05	0.06	0.18
Cl6-CB	13	0.16	0.07	0.15	0.29
Total CB		1.90	1.09	1.94	4.36
Cl2-CP	13	0.87	0.77	1.03	2.27
Cl3-CP	5	-	-	ND	0.10
Cl4-CP	7	0.09	0.10	0.04	0.30
Cl5-CP	10	0.47	0.52	0.18	1.66
Total CP		1.44	1.20	1.28	3.23

ND = less than the average Detection Level of 0.03 - 0.06 ng/m3 for PCB
= less than the average Detection Level of 0.03 ng/m3 for CB
= less than the average Detection Level of 0.03 ng/m3 for CP

acceptability level for hexachlorobenzene is 2 ng/m3 (9).

Mean total CP levels were 1.88 ng/m3 for the Windsor station, and 1.44 ng/m3 for the Walpole station. Pentachlorophenol and dichlorophenol accounted for over 80% of the total CP at Windsor, and for over 90% of the total CP at Walpole. Little ambient air data for chlorinated phenols has been reported in the literature. The province of Ontario guideline for an acceptable maximum 24 hour pentachlorophenol concentration is 20 ng/m3 (7).

INHALABLE PARTICULATE MATTER AND ASSOCIATED ELEMENTS AND IONS

PM10 data were available for 64 sampling days for the Windsor station, and for 23 sampling days for the Walpole station. A total of 40 elements plus sulphate and nitrate were measured on the fine (<2.5 µm) and coarse (2.5 µm to 10 µm) fraction of inhalable particulate matter collected with a dichotomous sampler. Tables 11 and 12 report the summary statistics for inhalable particulate matter, sulphate, nitrate and selected metallic elements for Windsor and Walpole respectively.

Mean PM10 concentrations were 36.8 µg/m3 for the Windsor station and 26.5 µg/m3 for the Walpole station. Of the metals measured, iron and zinc were found in the highest concentrations at both stations. Metals concentrations measured at the Walpole station were on average 3 to 5 times lower than those measured at the Windsor station.

The province of Ontario has air quality criteria for the metallic elements reported in Tables 11 and 12 (7). All values measured at the Windsor and Walpole stations were well below these criteria. The province has also collected a large historical data base on metals and particulates

TABLE 11. CONCENTRATION OF INHALABLE PARTICULATE MATTER (<10µm), SULPHATE, NITRATE AND SELECTED METALS - WINDSOR
(µg/m3)
64 SAMPLING DAYS

Compound	Detection Level	Mean(1)	Standard Deviation	Median	Maximum
Inhalable Particulate		36.800	21.900	31.000	102.000
Sulphate		6.680	6.420	4.370	29.550
Nitrate		1.240	1.420	0.850	9.380
Vanadium	0.0040	0.008	0.009	0.005	0.031
Chromium	0.0030	0.007	0.007	0.005	0.029
Manganese	0.0020	0.033	0.022	0.026	0.114
Iron	0.0020	0.729	0.563	0.518	2.477
Nickel	0.0010	0.003	0.002	0.003	0.010
Copper	0.0020	0.013	0.011	0.010	0.042
Zinc	0.0010	0.138	0.128	0.099	0.645
Arsenic	0.0005	0.001	0.001	0.001	0.004
Selenium	0.0005	0.003	0.002	0.003	0.007
Cadmium	0.0030	0.003	0.004	0.002	0.017
Antimony	0.0050	0.009	0.010	0.006	0.054
Lead	0.0008	0.068	0.029	0.063	0.145

(1) Concentrations are total of fine and coarse fractions
ND = not detected at Detection Level listed in Column 2

TABLE 12. CONCENTRATION OF INHALABLE PARTICULATE MATTER (<10µm), SULPHATE, NITRATE AND SELECTED METALS - WALPOLE (µg/m3) 23 SAMPLING DAYS

Compound	Detection Level	Mean(1)	Standard Deviation	Median	Maximum
Inhalable Particulate		27.000	18.000	23.000	74.000
Sulphate		4.670	5.510	3.230	24.890
Nitrate		0.560	0.410	0.550	1.470
Vanadium	0.0040	-	-	ND	0.013
Chromium	0.0030	-	-	ND	0.006
Manganese	0.0020	0.010	0.008	0.009	0.035
Iron	0.0020	0.232	0.217	0.174	0.915
Nickel	0.0010	-	-	ND	0.010
Copper	0.0020	0.007	0.006	0.005	0.016
Zinc	0.0010	0.028	0.022	0.016	0.090
Arsenic	0.0005	-	-	ND	0.002
Selenium	0.0005	0.002	0.001	0.002	0.004
Cadmium	0.0030	-	-	ND	0.003
Antimony	0.0050	-	-	ND	0.027
Lead	0.0008	0.017	0.011	0.014	0.052

(1) Concentrations are total of fine and coarse fractions
ND = not detected at Detection Level listed in Column 2

for Windsor and other urban centres in Ontario. Table 13 presents a comparison of concentrations of sulphate, nitrate and metallic elements measured in 1985 in Windsor and Sarnia with concentrations of the same elements measured at the Windsor and Walpole stations (11)(12). The Windsor results are comparable to the other urban results. The Walpole results are less than the urban results, by a factor of 2 or more.

TABLE 13. COMPARISON OF HISTORICAL DATA ON METALLIC ELEMENTS, SULPHATE AND NITRATE IN WINDSOR AND SARNIA (µg/m3)

Compound	Windsor MOE (1)	Sarnia MOE (1)	Windsor Station	Walpole Station
Sulphate	10.260	7.0700	6.680	4.670
Nitrate	4.700	2.9600	1.240	0.560
Vanadium	0.010	0.0100	0.008	-
Chromium	0.010	0.0060	0.007	-
Manganese	0.082	0.0211	0.033	0.010
Iron	2.178	0.5220	0.729	0.232
Nickel	0.008	0.0060	0.003	-
Copper	0.129	0.2900	0.013	0.007
Cadmium	0.004	0.0010	0.003	-
Lead	0.180	0.1110	0.069	0.017

(1) Average Concentrations from 1985 data across the entire urban area (network of 7 stations for Windsor, 4 stations for Sarnia)

QUALITY OF DATA

The sampling and analytical protocols employed in this study resulted in the collection of very high quality data. Blank concentrations were very low, recovery efficiencies for spiked surrogates were high and detection levels were very low for all compounds of interest.

FUTURE ACTIVITIES

The network will continue to operate, collecting additional data to add to the existing data base. Once the GDRRA Incinerator commences full operation, the network will begin collecting data which will be analyzed to determine if any significant increases in target parameters have been measured. A detailed air pollution emission inventory and source characterization will be necessary in order to properly assess the post-startup network data. This work will commence in the Spring of 1989.

Environment Canada anticipates the expansion of certain aspects of the Windsor/Walpole Ambient Air Monitoring Network throughout the province of Ontario, with the establishment, in co-operation with the Ontario Ministry of Environment, of a number of VOC and PAH monitoring stations in urban centres.

The International Joint Commission has been approached by both the Canadian and American governments regarding a reference on the ambient air quality in the Windsor/Detroit-Sarnia/Port Huron corridor. At the time this paper is being written, the Commission is still considering the request, so the format of the investigation under the reference is as yet unknown. However, when the Commission begins its work, the data collected by the Windsor/Walpole Network will form the basis for further investigation of this matter.

Environment Canada also looks forward to continued international co-operation and information dissemination through its ongoing participation in the Ad Hoc Committee for Ambient Air Monitoring for the Detroit Incinerator.

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PLANNING FOR AIR TOXICS MONITORING FOR MSW & RDF COMBUSTION FACILITIES

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During the past year WESTON in conjunction with Wayne County Air Pollution Control Division and Environment Canada have been involved in a comprehensive ambient air toxics monitoring program for a resource recovery facility (RRF). A monitoring protocol was developed to determine the actual effect of the RRF facility emissions on the surrounding environs. The protocol mandated one year of preconstruction monitoring to establish the background levels. After the RRF is operational, one additional year of monitoring will be conducted. A statistical analysis of sampling results from the two periods will be conducted to assess the impact of the new facility. The specific air toxic components investigated include, dioxin/furans, polychlorinated biphenyls, polyaromatic hydrocarbons, metals and inorganic acids.

The establishment of the method specific information for sampling and analysis, equipment, and advantages and disadvantages for measurement of each class of pollutants will be discussed. The development of quality assurance program will be reviewed.

INTRODUCTION

Ambient air monitoring programs for refuse derived fuel (RDF) facilities for municipal solid waste (MSW) require careful planning and design. The programs must answer the concerns of regulatory agencies and the community with sound data and yet maintain a reasonable cost for the operating authority. The precise definition of program objectives and the selection of sampling methodology and sample collection frequency are effective means to achieve this balance.

The ambient air monitoring project that has been initiated for the Greater Detroit Resource Recovery Authority (GDRRA) may be the most comprehensive program in North America for a resource recovery facility. This project involves sample collection and analyses conducted by Roy F. Weston, Inc. (WESTON) for GDRRA at a selected impact site for dioxin/furans (PCDD/PCDF), polynuclear aromatic hydrocarbons (PAH), polychlorinated biphenyls (PCBs) and related compounds, PM-10 trace metals (metals collected with particulate having an aerodynamic diameter less than 10 um), and acid gases. Meteorological parameters including precipitation are also collected at this location. Additional background monitoring is being conducted for semi-volatiles (PCBs, PCDD/PCDF and PAH) at a remote background site. Monitoring for semi-volatiles is being conducted by Environment Canada in Ontario, Canada at two additional sites.

GREATER DETROIT RESOURCE RECOVERY FACILITY

GDRRA's resource recovery facility located in Detroit, MI was constructed by Combustion Engineering Resource Recovery Systems. The facility is designed to receive and process a maximum of 4000 tons/day of MSW. Typically, the processing equipment will shred, screen and magnetically separate 3300 tons of solid waste to produce RDF, recyclable ferrous materials and nonferrous noncombustible waste five days a week. Each day, seven days a week, approximately 2400 tons of RDF will be used as fuel in the three spreader-stoker boilers for production of steam and electricity. Particulate emissions from each boiler will be controlled by a separate five stage electrostatic precipitator. Each precipitator will exhaust through a separate flue in a single stack. The construction was essentially complete early in 1989.

DEVELOPMENT OF THE SAMPLING PROGRAM OBJECTIVES

The environmental impact statement for this facility was published in February 1983 and subsequent addendums were added in August 1984. This study concluded that no adverse impact on ambient air quality would result from the operation of this facility. The Michigan Department of Natural Resources and the Wayne County Health Department granted construction permits on October 16,1984 and October 18, 1984, respectively. The air monitoring studies to determine the pollutant impacts on the ambient environment and the associated risk assessments relied upon assumed (albeit conservative) values. In light of continued concern over long term exposure to potentially hazardous emissions, GDRRA decided to measure the actual impacts of the air pollutants of concern.

INTERAGENCY COMMITTEE

To implement this decision, an ad-hoc interagency committee was formed in February 1987. The permanent members of this committee included the following groups:

- Wayne County Health Department - Air Pollution Control Division
- Michigan Department of Natural Resources
- Environment Canada - Toronto
- Environment Canada - Ottawa

- Ontario Ministry of the Environment
- GDRRA
- WESTON

The Wayne County Health Department acted as the focal point and coordinator of committee activities. Additional participants included the USEPA Environmental Monitoring Systems Laboratory and the City of Windsor, Ontario.

Meetings were conducted between March through July 1987 to forge a consensus on sampling and analytical methods, quality assurance and other elements of the ambient air monitoring program.

PROGRAM OBJECTIVES

The interagency committee concluded the objectives of the GDRRA study as follows:

- Document the back ground levels of pollutants of concern.
- Gather sufficient data to determine the actual impact of the GDRRA emissions on the environment.
- Integrate the applicable data from parallel monitoring programs.

Pollutants of Concern

While there are many pollutants of concern in ambient air, the study focuses on those pollutants that are specifically thought to be associated with combustion of municipal solid waste. Table 1 provides a list of pollutants that are included in the monitoring study.

The semi-volatile organics analyzed from the PAH, chlorobenzene, chlorophenol and PCB samples are presented in Table 2(1)

Determination of Data Sufficiency to Measure Impact

A statistical approach was selected to determine the level of sampling required to measure the impact on the ambient air quality. The study is split into two phases. The pre-construction phase measured background levels during the construction of the facility before the start of RDF combustion to establish background levels. The post-construction phase to monitor actual impacts continues the sampling for one year after operations have begun impacts. In the planning stage, an estimate of the number of samples required during each phase by using available data to calculate a standard

TABLE 1. LIST OF POLLUTANTS THAT ARE INCLUDED IN THE MONITORING STUDY

EMISSION	SYMBOL
Dioxins and Furans	
Polychlorinated dibenzo-p-dioxins and Polychlorinated dibenzofurans	PCDD,PCDF
Semi-Volatile Organics	
Polynuclear Aromatic Hydrocarbons	PAH
Polychlorinated Biphenyls	PCB
Chlorobenzenes and Chlorophenols	ClB, ClP
Trace Metals	
Antimony	Sb
Arsenic	As
Cadmium	Cd
Chromium	Cr
Copper	Cu
Lead	Pb
Manganese	Mn
Mercury	Hg
Nickel	Ni
Selenium	Se
Vanadium	V
Zinc	Zn
Inhalable Particulate Matter	
Inhalable Particulates	PM10
Acid Gases	
Hydrogen Chloride and Inorganic Acids	HCl

TABLE 2. SEMI-VOLATILE ORGANIC MEASUREMENTS

POLYNUCLEAR AROMATIC HYDROCARBONS (PAHs)

Acenaphtylene
Acenaphthene
Fluorene
Phenanthrene
Anthracene
Fluoranthene
Pyrene
Chrysene

Benzo(a)anthracene
Benzo(b)fluoranthene
Benzo(k)fluoranthene
Benzo(a)pyrene
Indeno(1,2,3-cd) pyrene
Dibenz(a,h)anthracene
Benzo(g,h,i)perylene

POLYCHLORINATED BIPHENYLS (PCBs)

10 Isomer Groups

CHLOROBENZENES AND CHLOROPHENOLS (ClB AND ClP)

Chlorobenzene
1,2-Dichlorobenzene
1,4-Dichlorobenzene
1,3-Dichlorobenzene
1,2,4-Trichlorobenzene

2-Chlorophenol
Pentachlorophenol
2,4-Dichlorophenol
Phenol
4-Chloro-3-methylphenol
2,4,6-Trichlorophenol
2,4,5-Trichlorophenol

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normal distribution for the expected results. This estimate of the number of samples was soon superceded by other practical considerations of the program design, particularly coordination with other sampling programs and the need to monitor over a range of meteorological conditions. These considerations tended to increase the required number of samples well above the initial estimates.

The actual determination of impact will be conducted as multilevel analysis of variance to consider site-specific effects as well as well as time series effects and correlations across pollutant data sets.

Integration of Parallel Sampling Programs

Several existing or proposed ambient sampling programs were considered valuable for integration with the GDRRA study. Environment Canada was proceeding with its own ambient monitoring program for semi-volatile organics to study the effects of the GDRRA facility on air quality. Priority pollutant (TSP,SO_2, lead, PM-10 and HCl) measurements were generally available through existing ambient air quality monitoring network maintained by the Wayne County Health Department. The EPA Urban Air Toxics program selected two sites in the Wayne County network for sample collection. These air toxic samples provide trace VOC data to the GDRRA program. The primary coordinating effort was made between the Canadian and GDRRA program sampling methodology and sampling schedules for PCBs, PNA and PCDD/PCDF.

Final Program Design

The final program design called for PAH, metals and acid samples to be collected every sixth day. PCB's and chlorobenzenes and chlorophenols are sampled every twelfth day. A dioxin/furan sample is collected every 24 days. To collect sufficient air sample volume to meet the required detection limits, semi-volatiles (PAH, PCBs and PCDD/PCDF) samples are collected over 48 hour periods. Other pollutants are collected over 24 hour periods. The 48 hours samples are collected from noon to noon. The 24 hour samples are collected from midnight to midnights. The ambient monitoring detection limits are presented in Table 3.(1)

TABLE 3. GDRRA AMBIENT MONITORING METHODS AND DETECTION LIMITS

COMPOUND	SAMPLING METHOD	ANALYTICAL METHOD	SAMPLE VOLUME	TARGET DETECTION LEVEL	SAMPLER
Dioxin/Furan	Hi-Vol/PUF	GC-MS	800 m3	0.05-0.5 pg/m3	General Metal Works PS-1
PCB	Hi-Vol/PUF	GC-MS	800 m3	2 - 20 pg/m3	General Metal Works PS-1
Chlorobenzenes	Hi-Vol/PUF	GC-MS	800 m3	10 ng/m3	General Metal Works PS-1
Chlorophenols	Hi-Vol/PUF	GC-MS	800 m3	10 - 50 ng/m3	General Metal Works PS-1
PAH	Hi-Vol/PUF-XAD	GC-MS-SIM	700 m3	0.02 - 0.2 ng/m3	General Metal Works PS-1
Particulates (PM10)	Med-Flow	Gravimetric	160 m3	---	Sierra Anderson Sampler Medium Flow
Metals	Med-Flow-TFE	XRF	160 m3	0.09 - 9 ng/m3	Sierra Anderson Sampler Medium Flow
Inorganic Acids	NIOSH 7903	IC Scan	0.45 m3	2 - 10 ug/m3	Silica gel Tube
VOC	EPA TO-14	GC-MS	16 L	0.1 ug/m3	Evacuated Canister

NOTES:

VOC samples are collected by Wayne County at the Playground Site every 12th day.

Sierra Anderson Medium Flow PM10 Samplers (4 CFM) with TFE filters are used for particulates and metals.

PCB, chlorobenzenes and chlorophenols are analyzed from the same sample.

IMPLEMENTATION OF THE MONITORING PLAN

Concurrent with the development of the program design, an evaluation of available sampling and analytical procedures was conducted. Although presented as a separate discussion here, the two processes were interactive. The intent was to ensure that the goals of the sampling program did not exceed the capabilities of the sampling and analytical procedures.

SITE SELECTION

Existing meteorological data was used to conduct a preliminary modeling study for the facility. Based on these results two sites were chosen for monitoring. The impact site is approximately one mile from the facility. The preliminary modeling predicted the greatest impact from the facility at this location. A background site several miles away which has similar urban ambient air conditions as the impact site was also chosen.

METEOROLOGICAL MONITORING

Meteorological monitoring is being performed at the impact site. Wind speed (scalar and vector), wind direction (scalar and vector), standard deviation of wind direction (sigma theta) which is a indication of atmospheric stability, temperature and precipitation are being collected at the 10 meter level. Prior to the design and installation of the meteorological station it is useful to check into the needs if the air dispersion and risk assessment models. All meteorological parameters being collected for the GDRRA study conform to PSD (prevention of significant deterioration) standards. The meteorological parameters being collected will be using in further modeling for the risk assessments. Precipitation data being collected is needed in the determination of wet/dry deposition calculations.

Meteorological data is being collected by a data logger which is the primary means of data archiving. The data prints out 5 minute and hourly averages hourly to the on site printer. Once daily all data is transmitted to the home office via PC modem relay from the data logger. In addition, data is also being collected by strip chart. This method of data collection allows for excellent data recovery barring any problems with the meteorological sensors. Multiple recording of the data, strip chart and data logger, allows for excellent QA of the data. The daily retrieval of data allows for daily QA and indicates any problems with data collection, retrieval systems and meteorological sensors.

There are a number of sources for meteorological equipment and data logging instruments. Climatronics, Weathermeasure, Campbell Scientific, ESC are just a few of the vendors that can provide meteorological instrumentation and data logging equipment. Prices for a PSD meteorological system depending on the options, range from 10,000-20,000 dollars for a single 10 meter level system. In addition to a regular schedule of calibrations and bi-annual audits need to be performed on each of the sensors to insure the performance remains within acceptable limits.(2)

INHALABLE PARTICULATE SAMPLING

The amount of airborne particulate for trace metals to be collected by the sampling protocol is determined by three primary requirements; 1) detection limit 2) sampler configuration 3) analytical technique, represent the need to obtain an accurate sample weight, and the need to sperate the sample into components of different size, density or composition. Particle collection may be for Total Suspended Particulate (TSP) or sampling determined by equivalent aerodynamic diameter (PM-10, particulate matter with a diameter less then 10 um) involving inertial separation of particles in to various fractions.

Sampling for TSP and PM-10 can both be accomplished by the use of high volume samplers which sample at rates of 40-60 cfm (1.1 to 1.7 m3/min). The PM-10 sampler uses inertial separation of particulate using a sampler inlet designed to provide the proper cut-point. Only particulate with a diameter less the 10 um in diameter is deposited on the filter. Medium flow PM-10 sampler operate in the same manner as the high volume sampler except the flow is in the range of 4 cfm (0.113 m3/min). Besides the lower flow rate the filter size is smaller.

Dichotomous samplers also measure particulate with a diameter of less the 10 um. A virtual impactor can separate particulate into course and fine fractions and deposit them on different filters. These samplers are generally considered low volume samplers, 0.56 cfm (1.67 x 10 -2 m3/min). The size selective samplers require a constant, controlled flow rate in order to maintain the inlet design criteria. Price of TSP samples are approximately 1,500 dollars and 3,500 to 7,000 for the size selective samplers. These sampler are available from General Metal Works and Wedding Associates.

The type of sampler used and the detection limits required will tend to dictate the size and type filter and the analytical technique used. High volume sampler requires the use of either paper (cellulose fiber) or glass fiber filters for good mechanical strength and low flow resistance. This is critical for the PM-10 samplers which need a constant flow rate. Teflon filters which have a high flow resistance may not be able to maintain the constant flow rate under heavy filter loading conditions in high volume samplers. Teflon filters are primarily used in the medium and low flow samplers.

TABLE 4. PARTICULATE PARAMETERS

Analysis Type	Cost (U.S. Dollars)	Detection Limit (@ 160 m^3 Samples)
XRF	90-150	0.09-9 ng/m^3
AA Flame(3)	90-150	1.25-250 ng/m^3
AA Graphite Tube(3)	285-300	0.05-6.25 ng/m^3

SEMI-VOLATILE SAMPLE COLLECTION

Measurement of semi-volatile compounds involve high volume sampling at flow rates from 300 to 1500 liter/min using filter media and sorbents. Sampling duration may be from 24 to 48 hours depending on the sample size needed to achieve the desired detection limit.

Alternative methods, such as cryogenic traps and impingers, have received very little use since these methods are not well suited for field applications. Continuous or quasi-real time methods can be applied to ambient air monitoring, but the high cost and narrow range of target compounds monitored, restrict the widespread applications of this technique. Continuous monitoring is still in the primitive state of development. Passive sampling techniques have been infrequently used for ambient area sampling, largely due to the long sampling times required because of the low concentration levels and limited analytical sensitivity.(6)

Particulate with aerodynamic equivalent diameters less than about 10 um are a special concern in the sampling program since these particles have a high probability of being deposited in the lung, and a size-specific reference method for this range has been promulgated. This method for semi-volatile sampling may be used in some instances instead of current high volume method or in conjunction with it. While PM-10 will provide valuable information on inhalable levels of semi-volatiles it will not yield total deposition which is used to determine other pathways of semi-volatile into to the body and other organisms.(6)

Several types of filters have been used to collect semi-volatile samples. Glass fiber, Quartz, Teflon, and Teflon-coated filters have the widest use. A comparison of regular glass fiber filters and Teflon coated filters have shown no difference in PAH catch from high volume sampling of ambient air. The quartz and glass fiber filter collected lesser amounts of PAH than Teflon filters.

Regardless of the type of filter used there is breakthrough of the shorter chain semi-volatiles. As a result other sorbents and filters, primarily PUF (poly-urethane foam) are used as a back-up. For PAHs, the sorbent XAD-2 resin is used in conjunction with the PUF to collect the short chain PAHs which in the vapor phase are not collected by the PUF alone.(6,7)

The integrity of material collected on the filters and sorbents during, and after sampling has been of concern in the monitoring of airborne pollutants. Chemical reactions of organic material collected on the filter with other pollutants such as NO2, O3, HNO3, SO2, H2SO4 as well as photooxidation and volatilization, especially during sampling, have been indicated as means by which sample integrity may be violated. These reactions reduce the quantity of the compounds available for analysis, thereby underestimating the actual ambient concentrations of pollutants. Therefore it is important to keep the sampling period as short as possible and recover the sample from the field as soon as sampling is completed. The same urgency is needed in returning the sample to the laboratory for extraction. The extracted sample is more stable and the final analysis can be conducted at a later time.(6)

There are a number of manufactures which produce PUF samplers. Prices of the sampler depend one the options, they generally range from 2,000 to 5,000 dollars from General Metal Works and Wedding Associates. They are available in either PM-10 and at various flow rates. Usually the high flow rate PUF samplers have large filters and PUF plugs. These larger samplers may require specialized glassware for the Soxhlet extraction. Some example values for various parameters of PAH sampling are presented on Table 5. Sample analytical cost will depend in part upon the number of samples submitted to the laboratory and the length of the sampling program. Detection limits are based are directly based on sample volume and analysis technique.

ANALYTICAL METHOD FOR PAH

Samples are usually extracted with organic solvents by using the SOXHLET extraction technique. The extract is then generally analyzed by GC/MS (gas chromatography/mass spectroscopy) in order to achieve the low detection limits required and to qualify and quantify the large variety of compounds. High resolution GC/MS may be used for the analysis of Dioxin/Furan and PCB and related compounds to achieve a lower detection limit at an increased cost. For PAH analysis in the selective ion monitoring mode using GC/MS will also reduce the detection limits and better qualify and quantify PAHs. This approach has been required in the GDRRA study, since the ambient air from urban areas tends to be "dirty", that is with high loadings of PAH from vehicle exhaust. The selective ion monitoring approach also increases the cost of the analyses.

TABLE 5. SEMI-VOLATILE PARAMETERS

Sample Type	Analysis	Cost (U.S. Dollars)	Sample Volume	Detection Limit
PAH	GC/MS	250-400	700	7.1-14.3 ng/m^3
	GC/MS SIM	800-1000		0.02-0.2 ng/m^3
Dioxin/ Furan	GC/MS	1000-1200	800	0.05-05 pg/m^3
PCB	GC/MS	800-1000	800	2-20 pg/m^3

QUALITY ASSURANCE

Quality assurance is an important part of any sampling program. The QA Plan should begin with the specifications established in the "Interim Guidelines and Specifications for Preparing Quality Assurance Project Plans" (U.S. EPA, 1980). A QA plan based on these guidelines should address the following items:

- QA objectives for each measurement parameter (precision, accuracy, completeness, reasonableness, and compatibility)
- sampling procedures.
- Sample custody.
- Calibration procedures and frequency.
- Analytical procedures.
- Data reduction, validation, and reporting.
- Internal quality control checks and frequency.
- Performance and system audits and frequency.
- Preventive maintenance procedures and schedule.
- Specific routine procedures to assess data precision, accuracy, and completeness.
- Corrective action.
- QA reports to management.

An important part of the GDRRA program has been duplicate samples using co-located samplers. Fully 25% of the semi-volatile PAH samples and 50% of the metals samples are duplicate co-located samples which are handled and analyzed in the same manner as ordinary samples. In addition field blanks are conducted for 15 % of all samples and trip blanks for each type of sample accompany each shipment to the laboratory.

For cross program quality assurance demonstration, WESTON samplers were set up at the Ontario sites and Canadian samplers were set up at the Detroit sites for parallel semi-volatile sampling. This feature was designed to determine the compatibility of sampling and analytical techniques. The comparability of data is an important consideration since all the data gathered during by the two independent sampling programs may be merged in to a single data base. The results of this side by side comparison have not yet been released.

Data reduction, validation and reporting on such a large quantity of data becomes important to insure that all the data collected is usable. This involves review of the data for consistency and reasonableness with other generated data, by the field technician during sampling, in the laboratory during analysis and finally when all parameters are combined to produce the final concentration in ambient air. This requires documentation by all personnel involved to any problems that occurred in the data collection process. Finally, a "usable" data base must be developed that permits easy access for data input in to various computer models.

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Hollywood, Florida USA
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ABSTRACT

The 1989 International Conference on Municipal Waste Combustion (MWC), sponsored by the United States Environmental Protection Agency and Environment Canada, is a special effort to provide an effective international forum for the transfer and exchange of technical information on municipal solid waste (MSW) incineration and ash treatment/disposal. Participating organizations include the American Society of Mechanical Engineers, the Air and Waste Management Association (formerly APCA), and the World Health Organization. The conference aims to provide an opportunity for professionals to meet and share ideas related to the planning, design, construction, operation, and maintenance of MWC facilities. This forum is also relevant to those involved in the regulation and research and development of new technologies addressing pollution problems related to MSW incineration.

The Conference Proceedings include papers examining relevant technical, regulatory, and social issues from the perspective of defining problems and proposing solutions, as well as describing experiences, forecasting trends, and reporting new and improved technologies. Papers and panel presentations are on the following topics: International Overviews; MWC Planning & Maintenance; Ash Characterization & Leaching; Risk & Health Effects; MWC Performance; Ash Treatment, Utilization, and Disposal; Sampling and Analysis; Residue Issues and Research Needs; Combustion Technology; Economic and Social Issues; Flue Gas Cleaning; Materials Recovery and Recycling; Quality Assurance/Quality Control, and Challenges of the '90's.

PREFACE

The International Conference on MWC, held on April 11-14, 1989, in Hollywood, Florida, is sponsored jointly by the United States Environmental Protection Agency and Environment Canada. The Conference is devoted to the exchange of current research developments on MSW incineration and ash disposal. Education in MWC and ash disposal and the dissemination of technical information are the primary objectives of the meeting.

Presented papers and two panel discussions comprise the program. The papers and panel presentations address a wide range and variety of MWC issues: International Overviews; MWC Planning & Maintenance; Ash Characterization & Leaching; Risk & Health Effects; MWC Performance; Ash Treatment, Utilization, and Disposal; Sampling and Analysis; Residue Issues and Research Needs; Combustion Technology; Economic and Social Issues; Flue Gas Cleaning; Materials: Recovery and Recycling; Quality Assurance/Quality Control, and Challenges of the '90's.

The conference speakers include Environment Canada and United States Environmental Protection Agency personnel, as well as representatives from federal and state environmental/health agencies, research and development groups, academic personnel, and manufacturers. Participants from Belgium, Denmark, Federal Republic of Germany, Italy, Japan, Sweden, Switzerland, The Netherlands, and United Kingdom add an international perspective on MWC research and development plus related issues.

The co-chairmen of the conference are Theodore G. Brna of the United States Environmental Protection Agency's Air and Energy Engineering Research Laboratory and Raymond K. Klicius of Environment Canada's Industrial Programs Branch, Conservation & Protection.

The Conference Proceedings are presented in two volumes and include papers received by March 14, 1989. Also, included are abstracts of papers which will not be available until the conference. The contents of each volume are organized as follows:

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of Mayors, Washington, DC
M. G. Johnston, Office of Air Quality
Planning and Standards,
U.S. Environmental Protection Agency,
Research Triangle Park, NC

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Morovia, CA
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"Utilization of Flue-Gas Cleaning Technologies for Waste Incinerators in Europe," W. Ellison 7C-43

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R. E. Sommerlad, Midwest Research
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Richmond, VA

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Agency, Research Triangle Park, NC

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Institute, Falls Church, VA
A. Finkelstein, Industrial
Programs Branch, Environmental
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U. S. Environmental Protection
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K. S. Felsvang, A/S Niro Atomizer,
Copenhagen, Denmark

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Concord Scientific,
Toronto, Ont.
S. Kulkarni, Center for
Environmental Quality,
Research Triangle Institute,
Research Triangle Park, NC

"Recommended QA/QC Activities for Municipal Waste Combustor Testing," C. E. Riley, W. G. DeWees, S. C. Steinsberger, R. R. Segall, and D. L. M. Dougherty 11A-1

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Session Co-Chairmen:
G. Dorian, Office of
Environmental Engineering and
Technology Demonstration,
U. S. Environmental Protection
Agency, Washington, DC
T. T. Eighmy, University of
New Hampshire, Durham, NH

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D. S. Beachler, Westinghouse
Electric Corp., Pittsburgh, PA
W. Ellison, Ellison Consultants,
Monrovia, MD

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THE DETERMINATION OF THE THERMAL OPERATING CHARACTERISTICS IN THE FURNACE OF A REFUSE-FIRED POWER BOILER

by: S. A. Scavuzzo and J. R. Strempek
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Wheelabrator Environmental Systems Inc.
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ABSTRACT

Recent concerns over the emission of trace organic compounds from solid waste incineration have led to governmental regulations affecting the design, operation, and maintenance of Refuse-to-Energy facilities. Many state environmental agencies are now implementing time-at-temperature criteria into their operating permits. These criteria vary from state to state but are based on the belief that the destruction of organic compounds is a function of furnace gas residence time-at-temperature in the presence of oxygen in a well-mixed state. Extensive testing is often required to verify permit compliance capabilities and/or to develop correlations for on-line continuous monitoring. This paper discusses the procedure and equipment used to perform time-at-temperature testing at facilities located in Millbury, MA., and Bridgeport, CT., and covers typical results of such analyses.

INTRODUCTION

The passage of Section 102 of the Hazardous and Solid Waste Amendments (HSWA) of 1984 was a response to a new level of public concern over the emission of trace organic compounds from municipal waste combustors. This act mandated the Environmental Protection Agency (EPA) to prepare and submit to Congress a report setting forth the combustor design criteria and operating practices deemed appropriate for controlling these emissions(1). Anticipating this legislation, state permitting agencies began to specify various measurable quantities which, on a design basis, were thought to represent 'good combustion practices'. Most of these criteria included considerations for providing a high combustion gas temperature (1500 - 1800°F) for a sufficient length of time (1 - 3 seconds). It was presumed that a combustor designed to these requirements would be capable of operating at high combustion efficiencies.

As a condition of the operating permits for the 1500 tons per day (TPD) Millbury, MA., and 2250 TPD Bridgeport, CT., facilities, it was necessary to predict that the furnaces would meet certain time-at-temperature criteria on a theoretical basis and to demonstrate compliance to these criteria in actual field tests. A detailed parametric engineering study was concluded in 1985, and the field tests were conducted from December 1987 to June 1988.

BACKGROUND

The units tested at Millbury and Bridgeport are two identical balanced-draft Babcock & Wilcox Stirling Power Boilers integrated with Von Roll refuse grate systems (Figure 1). Each unit utilizes a mass-burn water-wall technology designed to process 750 TPD of Municipal Solid Waste (MSW) and to generate 192,000 lb/hr of superheated steam at conditions of 900 psig and 830°F. Each unit is top supported and is arranged with water-cooled furnace walls of gas-tight membrane construction, a three-pass pendant superheater, a modular generating bank, and a modular economizer. The boilers are an integral part of each facility operated by Wheelabrator Environmental Systems Inc. (Figure 2).

In the combustion process, refuse fuel is converted to heat, gas, and residue by a combination of the "three T's" of combustion: time, temperature, and turbulence(2). The extent to which combustion may be optimized is a function of the mechanical ability of the combustion system to mix air with fuel. In this case, the combustion system is comprised of the refuse grate system, the air system, the control system, and the furnace(3). Fuel and air are admitted in a controlled manner by which heat and gaseous products are liberated. As these hot gaseous products pass through the furnace, a portion of their heat is transferred to the surrounding furnace water-wall enclosure. Though control system adjustments can be made to the refuse grate and air systems during operation to optimize combustion conditions, no operational adjustments can be made to the fixed furnace walls. Therefore, the as-built furnace must be capable of successfully

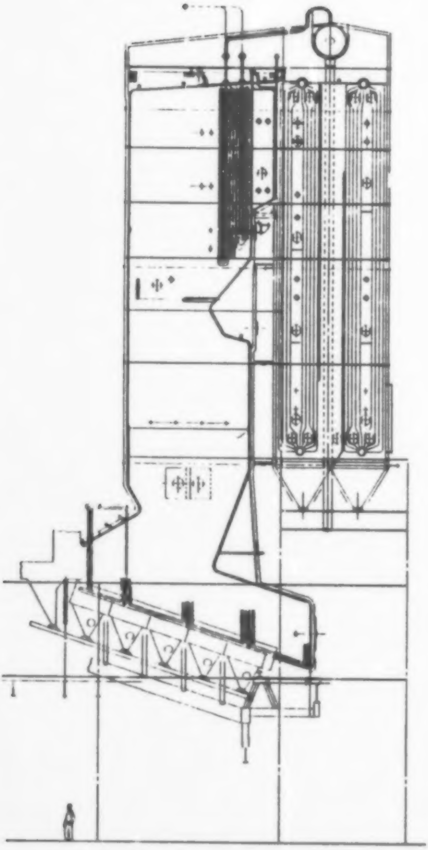

Figure 1 *Side View of a B&W 750 TPD Mass-Fired Power Boiler.*

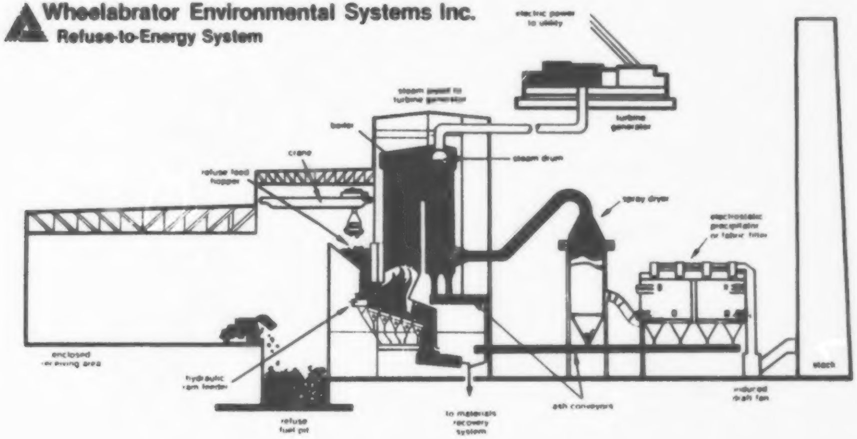
Wheelabrator Environmental Systems Inc.
Refuse-to-Energy System
electric power to utility
steam piped to turbine generator
turbine generator
boiler
crane
steam drum
refuse feed hopper
spray dryer
electrostatic precipitator or fabric filter
enclosed receiving area
hydraulic ram feeder
refuse fuel pit
to materials recovery system
ash conveyors
induced draft fan
stack

Figure 2 *Test Facility Operated by Wheelabrator Environmental Systems Inc.*

handling a wide range of operating conditions(4).

For a given size furnace, the velocity of the gaseous combustion products passing through it is mainly determined by the flow rate of gas present. In turn, the velocity of the gas has a profound effect on:

a) aerodynamic suspension and transport of particulate matter, and
b) the time available for the gas to reside in the furnace at a temperature sufficient to complete combustion.

The concept of furnace gas residence time-at-temperature is described by the simultaneous liberation of heat from the combustion of the refuse and the absorption of heat by the furnace water-walls as the gas moves through the furnace. The combined effects of conduction, convection and radiation heat transfer within the furnace, together with the energy associated with gas mass transfer, can be used to predict the temperature and location of a furnace gas control volume at any given moment in time. When considered over the entire furnace volume, this analysis can be used to generate a thermal operating characteristic, more commonly referred to as a time/temperature curve. This curve provides the design engineer with a basis for evaluating the adequacy of a particular furnace relative to its intended refuse handling capacity(5).

THEORETICAL PREDICTION OF RESIDENCE TIME-AT-TEMPERATURE

The methodology used to predict furnace gas residence time-at-temperature relies upon field-proven methods to determine furnace absorption rates for bare tube and refractory covered enclosure surfaces. Adjustments are incorporated to account for some ash coating. The furnace model is divided into zones to account for the fuel and air introduction locations as well as refractory versus bare tube surface. A computer model is used to calculate temperature, quantity, and composition of gas entering and leaving each zone from the given fuel analysis and excess air. To simplify the model for the complex natural gradients that occur in actual practice, gas temperature and flow are considered to be evenly distributed at each zone boundary. The volumetric gas flow rate for each zone is then determined from gas weight and density at the average temperature of the zone. Finally, the zone residence time is calculated by dividing zone volume by the volumetric gas flow rate.

The residence time calculation for each zone in the furnace is as follows:

$$RT_i = \frac{V_i}{G_i} \qquad (1)$$

where: RT_i = Residence time in zone i, seconds.
V_i = Volume of zone i, cubic feet.
G_i = Volumetric gas flow rate through zone i, cubic feet per second.

$$Tavg_i = \frac{(T1_i + T2_i)}{2} \tag{2}$$

where: $Tavg_i$ = Average temperature in zone i, °F.
$T1_i$ = Temperature entering zone i, °F.
$T2_i$ = Temperature leaving zone i, °F.

The volumetric gas weight, G_i, can then be determined from:

$$G_i = \frac{Wg_i}{\rho_i * 3600} \tag{3}$$

where: Wg_i = Gas weight entering zone i, lb/hr.
ρ_i = Gas density at average zone temperature $Tavg_i$, lb/ft^3.
3600 = Conversion, sec/hr.

By substituting, equation (1) can be rewritten as:

$$RT_i = \frac{V_i * \rho_i * 3600}{Wg_i} \tag{4}$$

The cumulative residence time in the furnace (RT) at any point, is the algebraic sum of the residence times of the preceding zones, or:

$$RT = \Sigma_{i=1}^{n} \; RT_i \tag{5}$$

where: n is the number of furnace zones.

The time/temperature curve is then derived by plotting the gas temperature and cumulative residence time with the corresponding location of each zone boundary.

FIELD MEASUREMENT OF RESIDENCE TIME-AT-TEMPERATURE

FURNACE CONDITIONS

Prior to any testing, the units were commissioned in accordance with the manufacturer's recommendations and run at full load firing refuse for several weeks to allow the heat absorbing surfaces to attain 'commercially clean' conditions. During this time, the automatic combustion controls were tuned and the required plant instrumentation was calibrated.

All tests were conducted at steady state conditions with automatic controls holding a steam flow setpoint. No sootblowers, rappers, or auxiliary burners were operated during the tests.

TEST EQUIPMENT

To obtain on-line furnace time/temperature results, an automated data acquisition and analysis system was employed. A data acquisition computer was interfaced with all temporary test instrumentation, the plant's Network 90®[1] data highway, and the furnace temperature measuring system to provide real-time access to all necessary instrumentation. Sensors were scanned on a 15-to-60 second time interval, depending on the transient nature of the process, to capture enough data to measure the natural transients that occur in the unit. Multiple scans were then averaged over 10-minute-time intervals and the data were reduced to develop on-line residence time/temperature curves. The curves were then displayed on the CRT of the data analysis computer. All test data were also archived for future use.

FURNACE TEMPERATURE MEASUREMENT

A necessary parameter for the calculation of furnace residence time-at-temperature is the temperature profile in the combustion zone. It is also one of the most difficult parameters to measure due to the dynamic nature of the combustion process. Historically, water-cooled high velocity thermocouple (HVT) probes have been used to measure gas temperature in high temperature zones. The disadvantages of HVT traverses are that they are manpower-intensive; they take a long time to complete; and they give single-point readings which are often not representative of the conditions during the entire traverse period.

To satisfy the need for instantaneous furnace temperature measurement at multiple locations, a combination of optical pyrometry and acoustic pyrometry was employed.

Acoustic pyrometry is a technique for determining gas temperature based on the propagation speed of acoustic waves(6). For a given gas composition, the propagation speed varies as the square root of the absolute temperature. The basic objective of this technique is to generate and transmit a unique pattern of acoustic waves, detect their arrival, and accurately measure the transmitter-to-receiver flight time. By knowing the transmission path length and flight time, an acoustic velocity is computed. The specific heat ratio and molecular weight of the flue gas are then used to determine the average temperature along the propagation path.

B&W's Pyrosonic 2000®[2], which employs the acoustic techniques already described, was installed to measure the furnace gas temperature at six locations on five separate elevations. The measurement locations were selected to avoid regions of stratified gas/air which would otherwise have had an adverse affect on the results. HVT traverses were used to verify

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[1] Network 90 is a registered trademark of The Bailey Controls Co.

[2] Pyrosonic 2000 is a registered trademark of The Babcock & Wilcox Co.

Pyrosonic 2000 at all locations prior to testing. A two-color optical pyrometer was used to obtain the temperature just above the grate because test ports for installation of Pyrosonic 2000 were not available at this location. Figure 3 shows the location of gas temperature measurements made using the three techniques already described.

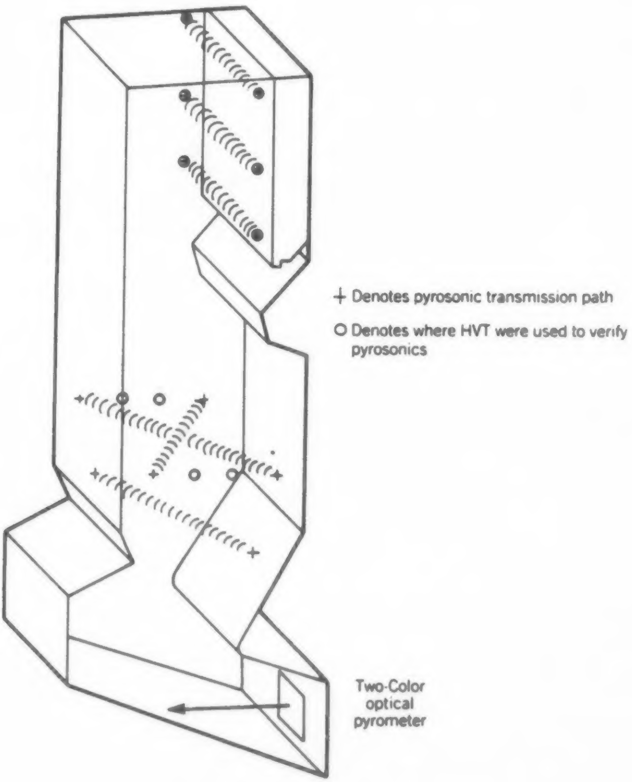
+ Denotes pyrosonic transmission path
O Denotes where HVT were used to verify pyrosonics
Two-Color optical pyrometer

Figure 3 *Pyrosonic locations*

FLUE GAS FLOW MEASUREMENT

The actual flue gas flow was measured on-line by a grid of probes designed to measure static and total pressure (Figure 4). The grid was calibrated by performing manual flow traverses of the flue over the practical load range of the boiler to develop a calibration factor. This factor was used to relate the velocity pressure from the grid to the actual flue gas flow rate. The manual flow traverses were done in general accordance with ASME PTC 11 Fans, using a three-hole, calibrated Fechheimer probe.

FLUE GAS MOISTURE MEASUREMENT

Flue gas moisture was obtained in accordance with EPA methods 1, 2, and 4 by traversing the economizer outlet/spray dryer absorber (SDA) inlet flue (reference Figure 4 for test port locations). The moisture content measured during each test was used for the final calculation of flue gas flow and as a variable input to Pyrosonic 2000.

FLUE GAS ANALYSIS

A multipoint sampling grid was installed at the economizer gas outlet in general accordance with ASME PTC 19.10. The grid was used to obtain a representative measurement of flue gas temperature and gaseous constituents.

The gas samples were extracted, mixed in composite bubblers, and continuously analyzed for dry volume percentages of O_2, CO_2, and CO. The resulting flue gas analysis was used as a variable input to Pyrosonic 2000; to calculate gas density; and to calculate the gas weight below the elevation of overfire air (OFA) injection. Figure 4 denotes the locations of the sampling grids used to determine gas constituents.

DETERMINATION OF RESIDENCE TIME / TEMPERATURE CURVES

The furnace was divided into zones as defined by location of temperature measurement devices and the location of OFA injection, as shown in Figure 5. The gas temperatures at the upper boundaries of zones 1 and 2 were interpolated from temperatures measured at the zero reference elevation and the upper boundary of zone 3. The zero reference elevation (0 feet) is defined as the elevation at which a vertical line drawn from the tip of the front wall arch intersects the grate. The reference elevation is arbitrary and has no effect on the results.

Because approximately 45% of the combustion air enters the furnace through the OFA ports, it is necessary to account for the lower gas weight below this elevation. The furnace gas weight, below OFA injection, was approximated by subtracting OFA from the measured gas weight at the boiler exit. The measured undergrate to overfire air ratio was obtained from the plant's permanent instrumentation.

The gas density was calculated using gas analysis from the economizer outlet and the average measured temperature in each of the furnace zones. The gas residence time in each zone was then determined using Equation (4).

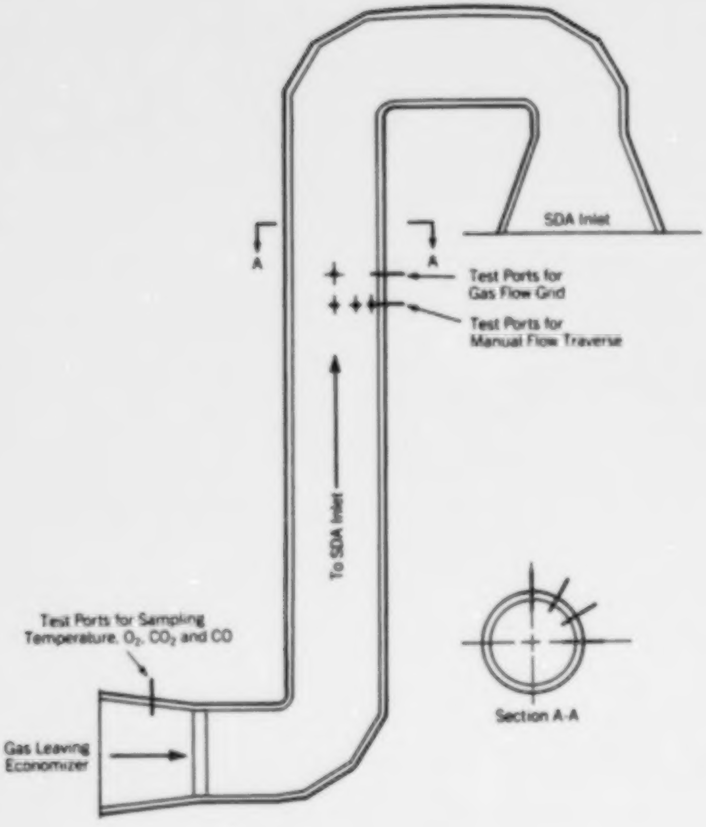
SDA Inlet
A
A
Test Ports for Gas Flow Grid
Test Ports for Manual Flow Traverse
To SDA Inlet
Test Ports for Sampling Temperature, O₂, CO₂ and CO
Gas Leaving Economizer
Section A-A

Figure 4 *Gas Sampling Locations at Economizer Outlet*

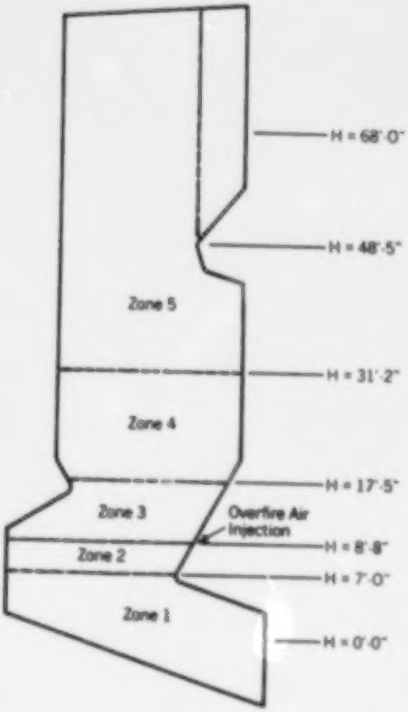
H = 68'-0"
H = 48'-5"
Zone 5
H = 31'-2"
Zone 4
H = 17'-5"
Zone 3
Overfire Air Injection
H = 8'-8"
Zone 2
H = 7'-0"
Zone 1
H = 0'-0"

Figure 5 *Furnace zones for residence time calculations*

PRESENTATION AND INTERPRETATION OF RESULTS

Steady state tests, of four-hour duration, were conducted at several boiler steam loads representing the practical operating range of the units. The data recorded during each of the test runs was averaged and reduced to develop time-at-temperature relationships. Test results obtained from both of the units were in close agreement.

Figure 6, produced from full-load test data recorded at the Bridgeport facility, is a graphical representation of the algorithm used to interpolate the gas temperature and cumulative residence time at any given elevation in the furnace. As an example, Figure 6 can be used to interpolate the residence time to be 1.9 seconds at an elevation of 20 feet. The corresponding gas temperature at 20 feet may then be read from the temperature ordinate.

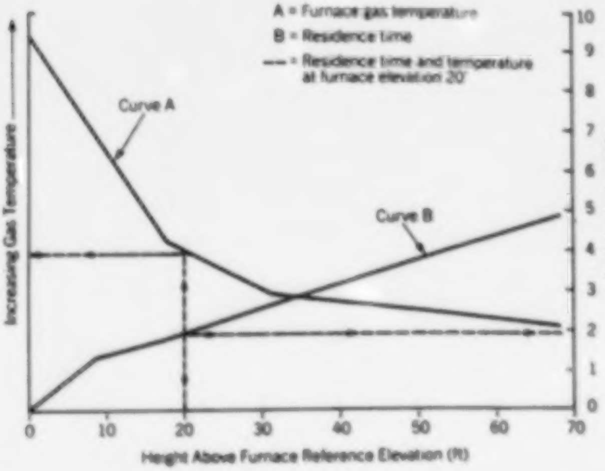

Figure 6 *Gas Temperature and Residence Time vs Furnace Height at Full Load*

The residence time and associated gas temperatures between any two elevations may also be determined with time/temperature curves. Figure 7 is identical to Figure 6 except that tutorial lines have been added to show how to interpolate the gas temperature one second after OFA injection. Beginning on the abscissa at 8.7 feet, follow a vertical path to the intersection of the residence time curve B. The value read from the residence time ordinate, 1.3 seconds, is the residence time at the elevation of OFA injection. Add one second to determine the time at one second after OFA injection, 2.3 seconds. Following a horizontal path back to the residence time curve B, the elevation at one second after OFA injection is determined, 27 feet. A vertical path is then followed until it intersects with the gas temperature curve A and the gas temperature one second after OFA injection may be read from the gas temperature ordinate.

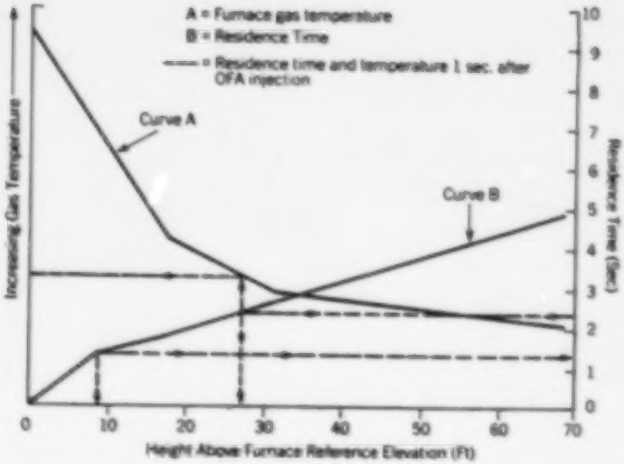

Figure 7 *Gas Temperature and Residence Time 1 Second After OFA Injection*

The test data show that the elevation of a given temperature plane within the furnace changes as boiler load and other factors change. Similarly, the gas temperature at a given residence time downstream of a reference point changes as boiler load and other factors change. To illustrate this point, reference Figure 6 to find the elevation of the plane where gas residence time is one second after OFA injection, 27 feet. Figure 8 is a plot of gas temperature and residence time at elevation 27 feet versus steam flow. Note that as unit load decreases, gas temperature at elevation 27 feet decreases as expected, and residence time at 27 feet increases substantially. This example clearly demonstrates that it is not possible to install a temperature monitor on the furnace wall at a position representing one second of residence time after OFA injection for all possible operating conditions.

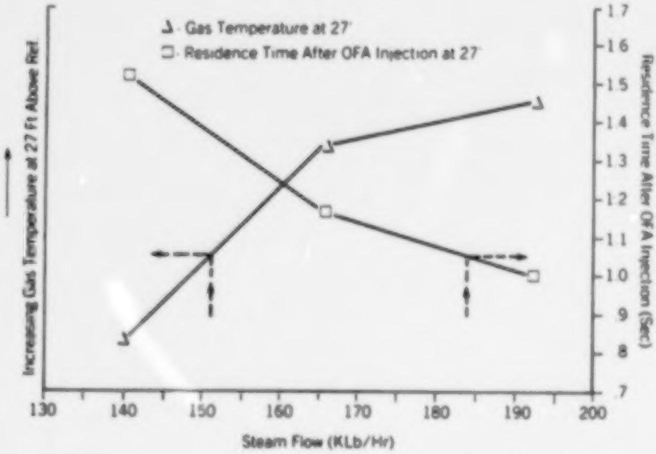

Figure 8 *Residence time after OFA @ 27' and gas temperature vs steam flow*

Figure 9 is a comparison of the gas temperature at one second residence time after OFA injection for two different load points. It is identical to Figures 5 and 6 except that the time/temperature curves developed from the low steam load test (140,000 lb/hr) have been included. Although it seems logical that the gas temperature at a given residence time should decrease with load, Figure 9 shows that the gas temperature at one second after OFA injection is actually higher at the low load condition than at the high load

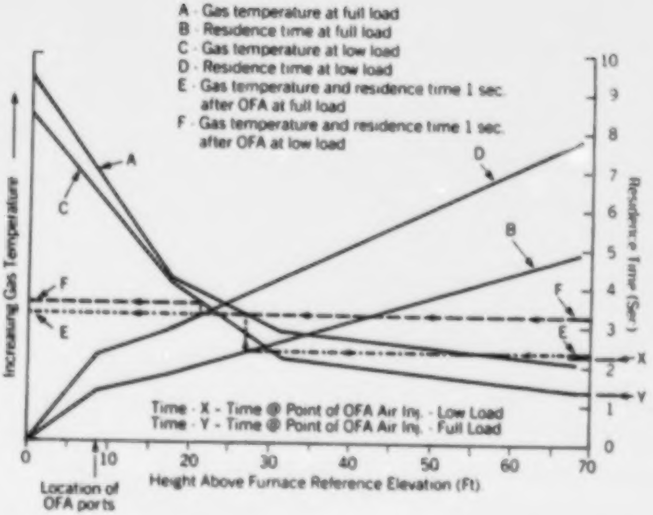

Figure 9 *Gas Temperature @ 1 Sec. Residence Time After OFA Injection vs Furnace Height*

condition. This occurs because at low load, the reduced gas weight has more of an effect on residence time-at-temperature than does the generally lower gas temperatures. This further illustrates the dependence of residence time-at-temperature on unit operating conditions.

In order to evaluate the influence of varying operating conditions on residence time-at-temperature, a second test was conducted at the full load condition presented in Figures 6, 7, and 9. This test was conducted several days later to allow for potential changes in fuel and furnace conditions; additionally, the unit was operated with approximately 25% lower excess air. Figure 10 is a plot of the time/temperature curves developed from the two full load tests which shows that both gas temperature and residence time were affected by changes in operating conditions.

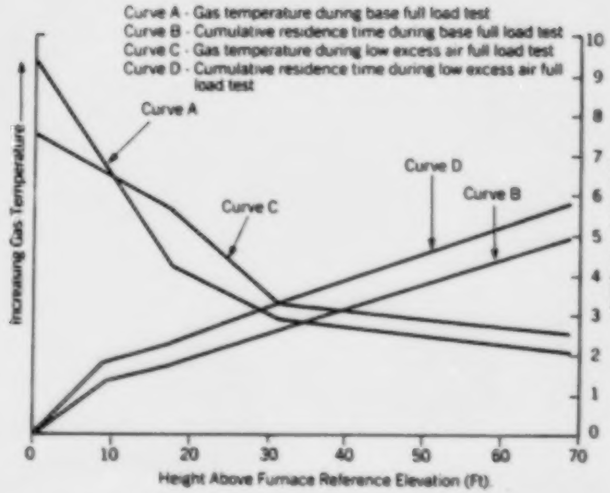

Figure 10 *Gas Temperature and Residence Time at Two Full Load Conditions.*

Based on the analysis of the Millbury and Bridgeport test data, the following is noted:

- The thermal operating characteristics of a refuse-fired power boiler can be monitored on-line.
- Acoustic pyrometry is an effective means of measuring instantaneous temperatures at multiple locations in the combustion zone.
- Both of the units tested satisfied the permit requirements for residence time-at-temperature.
- There is not a single location in the furnace representative of one second residence time for all operating conditions.
- Residence time-at-temperature is affected by a number of variables including unit load and excess air.

SUMMARY

The dynamic nature of flow and temperature conditions in the furnace of a refuse-fired power boiler require rigorous measurement and analytical techniques. The application of fundamental heat transfer and gas dynamics principles, together with state-of-the-art measurement and data acquisition, provides the technology to determine the actual thermal operating characteristics. This technology has been successfully used to demonstrate compliance with furnace gas residence time-at-temperature requirements at two commercial Refuse-to-Energy facilities. The test results confirm the theoretical basis used in the prediction of residence time-at-temperature, and will enable subsequent predictions to be made with greater confidence.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

ACKNOWLEDGEMENTS

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RETROFIT OF NATURAL GAS FIRING TECHNOLOGY FOR CONTROL OF CHLORINATED DIOXIN/FURANS IN EXISTING RDF COMBUSTORS

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ABSTRACT

The Gas Research Institute has been funding studies to develop gas fired technologies which can be retrofit to existing municipal solid waste combustion systems to control PCDD/PCDF emissions while providing other operational benefits. The initial application of this technology has been focused on Refuse Derived Fuels (RDF) fired combustion systems. The first studies were designed to characterize a full scale RDF furnace and has provided new insight about where and how PCDD/PCDF are formed. Pilot scale studies have been used in the development of the optimum design of the natural gas cofiring system. A full scale evaluation of the technology is currently underway.

INTRODUCTION

The Gas Research Institute is supporting the development of gas cofiring technologies for municipal waste combustion systems in order to control pollutant emissions and to improve other performance parameters. The initial focus of this activity has been on Refuse Derived Fuel (RDF) fired systems, but eventually, all municipal waste combustion technologies will be included. RDF units are unique among municipal waste combustion systems in that the waste is processed before it is fired. The processing includes size reduction accomplished by shredding and reduction of inorganic content accomplished by magnetic and physical separation. Typically, the waste is reduced to a top size between 4 and 6 inches, ferrous materials are removed magnetically and other high density inorganic materials are screened. The resulting RDF is then air blown into a water tube furnace and the heavy refraction falls onto a traveling grate where combustion is completed. As much as 50% of the RDF will be entrained in the furnace flow and never reach the grate. Thus, in RDF systems, there are two distinct modes of burning that occur; in suspension and on the grate.

Despite the fact that the waste is processed, general practice indicates that the burning characteristics of RDF in this mode tends to have some limiting problems. These problems arise due to the RDF feed instabilities and non-uniformities. These problems include(1,2,3,4):

- Poor load following characteristics
- Cold start limitations
- Organic emissions
- Corrosion of tube walls
- Slagging and fouling

The objective of this study is to evaluate gas cofiring technologies that will improve performance of RDF combustion systems by damping out the fluctuations associated with feeding and burning the hetrogeneous RDF fuel. The most pressing problem for plant owners and operators are environmental regulations. Increasingly stringent emission requirements are being placed on municipal waste combustion systems, in particular, standards will soon be proposed for polychlorinated dibenzo (P) dioxins and furans (PCDD/PCDF), HCl and particulate matter. There is also some interest in establishing NO_x standards for these units not only in the local noncompliance areas, but also on a national basis. It is unlikely that current RDF spreader stokers can meet the more stringent PCDD/PCDF emission standards that are being considered on a federal level (see Table 1) without modifications such as combustion improvements or the use of air pollution control devices. A gas-firing based system may provide destruction of PCDD/PCDF (not just removal with the flyash) as well as potentially other operational benefits such as efficient start-up and load following.

TECHNOLOGY GOALS FOR GAS COFIRING IN RDF SYSTEMS

A series of technology goals has been developed for gas cofiring which, if achieved, will enable the gas technologies to be incorporated into RDF technologies. Because the control of PCDD/PCDF represents the most substantial technical challenge it is the early focus of this study. The following are the selected requirements for environmental constraints that represent the goals of the gas cofiring technology:

TABLE 1. PCDD/PCDF EMISSIONS FROM RDF PLANTS

Unit Name	Type	Test Condition	APCD	PCDD/PCDF (ng/dscm) 12% CO2		2,3,7,8-TCDD Toxic Equiv (ng/dscm)		CO (ppmdv) 12% CO2		Partic (gr/dscf) 12% CO2	
				APCD Inlet	Stack	APCD Inlet	Stack	APCD Inlet	Stack	APCD Inlet	Stack
This Study: Run 1	RDF	Low Load	ESP	1237.1	260*	13.1		150			
Run 2B	RDF	Normal	ESP	532.5	130*	5.0		70		1.52	
Eastern RDF Unit #1	RDF	Normal	SD/FF	877		11.2				3.56	
Akron	RDF	Normal	ESP		1065						0.265
Albany	RDF	Normal	ESP		657		110		346		0.161
Hamilton Wentworth	RDF	F/back	ESP		13500		1660		430		0.257
Wright Patterson AFB	dRDF	Normal	Cyc./ESP		215		0.5				
Haverhill	RDF RDF	Normal Post Retro	ESP		3100 117		43				
Eastern RDF Unit #2	RDF	Normal	SD/FF	1000							
Niagra Falls	RDF	Normal Post Retro	ESP		2561 4246				254 204	3.25	

* Estimated from Input Gas Concentration and Ash Levels

• Median RDF Stack Emissions 440 ng/dscm

1) Emissions of PCDD/PCDF should be reduced to a maximum of 100 ng/dscm.

2) A minimum of 50% acid gas removal.

3) A one hour rolling average of CO of less than 100 PPM (corrected to 0% O_2)

4) Load following capability with an appropriate ramp up/ramp down speed with steam conditions in flow rate constant within ± 5%.

5) Smokeless rapid start-up (Opacity less than 20%, smoke number less than 2).

Gas cofiring should make MWC's meet these requirements and thus, represent the goals of the gas cofiring technology. In addition, the cofiring technology must be cost effective and retrofitable if it is to have general applicability. This implies the minimization of natural gas use (10-20% of total heat load depending on the configuration, gas cost, and alternative control technologies) and a method for low cost equipment installation. The retrofit costs should be less than $500,000 to ensure competitiveness. Finally, gas cofiring must not have adverse effects on other operational issues, such as slagging, fouling and corrosion.

PCDD/PCDF EMISSIONS FROM RDF COMBUSTORS

The causes of PCDD/PCDF emissions from RDF combustors is becoming more clear, but has not yet been resolved. The current theories concerning PCDD/PCDF emissions suggests that four specific formation and/or escape mechanisms may be responsible for emissions from MWC's(5). The importance of each mechanism may vary from unit type to unit type, but the predominant mechanisms include the following:

- Flame zone by-products wherein PCDD/PCDF is formed from precursors within the unburned regions of the flame.
- Down stream (approximately at 300°C) conversion of precursors catylized by fly ash.
- Escape of PCDD/PCDF volatilized from the waste containing PCDD/PCDF.
- Pyrolysis reactions occurring on entrained particles which escape the furnace unburned.

There is support and contradictory evidence for all of these in the extensive literature on PCDD/PCDF. Studies have been conducted on an RDF unit by GRI in order to define which of these units is dominate for this particular type of unit. These tests were also designed to provide a characterization of the conditions that exist in an RDF unit to which gas cofiring is to be applied. This RDF-fired facility consisted of two identical electrical generating boilers with a total combined electrical generating capacity of 18 MW. The boilers were originally designed to burn coal on a stoker, but were retrofit to allow the combustion of RDF. The modification which occurred in 1986 enlarged the boiler by extending the furnace 14 feet downward. A schematic of the current boiler configuration is provided in Figure 1. The configuration includes the boiler furnace an economizer, an air heater and an electrostatic precipitator. RDF is fed to the units by four pneumatic air swept fuel feeder spouts. RDF is fed to these spouts with two live bottom bins equipped with screw type feeding systems. The RDF is prepared at a remote processing plant from municipal refuse and trucked to the plant. The preparation includes select removal of bulky items such as refrigerators, and magnetic separation of ferrous materials. The shredding of the refuse is accomplished such that the Refuse Derived Fuel will pass through a 6 inch mess screen. More details concerning the operation and design of this unit is provided elsewhere(6).

A series of boiler characterization tests were conducted to establish baseline unit operating characteristics and pollutant emission levels. Tests were conducted under normal operating conditions, at two steam generation rates, and under modified operating conditions to investigate the sensitivity of PCDD/PCDF emissions to operating variables. Emphasis was placed upon variables thought to impact PCDD/PCDF formation including gas temperatures and residence time, combustion efficiency indicators and ash particle parameters. Boiler feed rates, air flow rates and system temperatures were recorded in an attempt to correlate these parameters with emissions levels.

Another objective of the test was to examine temporal and spatial operating non-uniformities in the furnace. Temperature profiles across the upper furnace (See Figure 1) were made to determine if any low temperature pathways existed by which furnace gases could pass through the boiler without being held at high temperatures. The test also measured carbon monoxide at various points across the upper furnace to determine if any regions having poor combustion efficiency existed. Furnace gas mixing and homogeneity were evaluated by measuring oxygen/gas concentration across the upper furnace. The uniformity of the flow field was examined by measuring velocity profiles across the upper furnace as well.

The key focus of the studies was to characterize emission levels from the unit and to evaluate the potential dioxin mechanisms. These measurements were designed to determine the regions in which the PCDD/PCDF were formed as well as temperatures, residence times and combustion conditions in these regions. Emissions data could then be compared to results from other MWC units and control methods be developed for the specific conditions of RDF combustion systems. As shown in Figure 1, carbon monoxide, particulate emissions and PCDD/PCDF emissions were simultaneously measured at two locations which encompassed the temperature window in which significant dioxin formation has been reported to occur(7-11). Ash samples were also obtained and analyzed for PCDD/PCDF as well as residual carbon, hydrogen and nitrogen.

Data from the full scale RDF unit suggest that particle matter carryover from the furnace is the most important single parameter which influences PCDD/PCDF emissions. In Figure 2, emissions data are shown from this unit along with the Quebec City mass burn unit plotted against the particulate matter

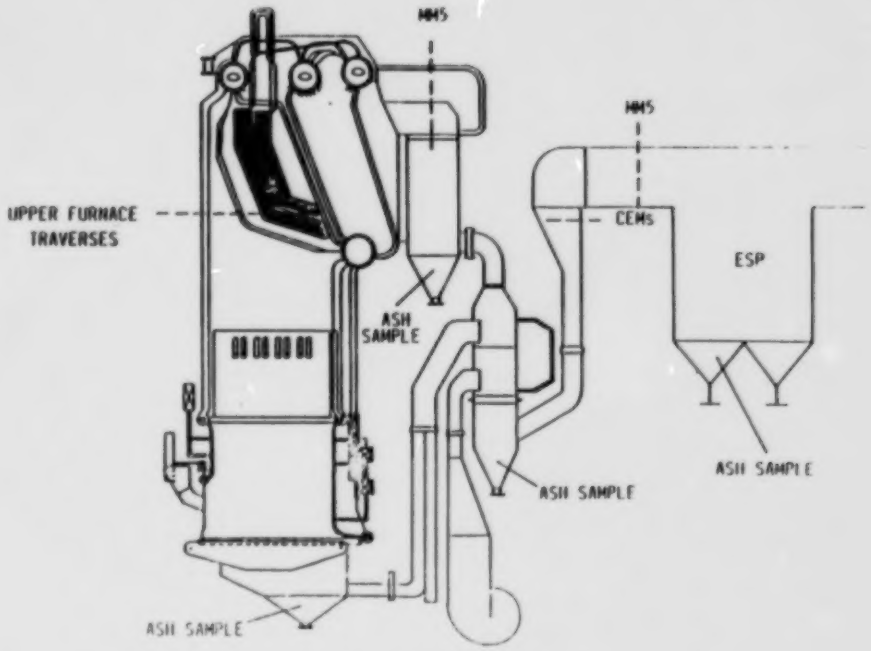

MM5
MM5
UPPER FURNACE TRAVERSES
CEMS
ESP
ASH SAMPLE
ASH SAMPLE
ASH SAMPLE
ASH SAMPLE

Figure 1. RDF Test Unit Schematic.

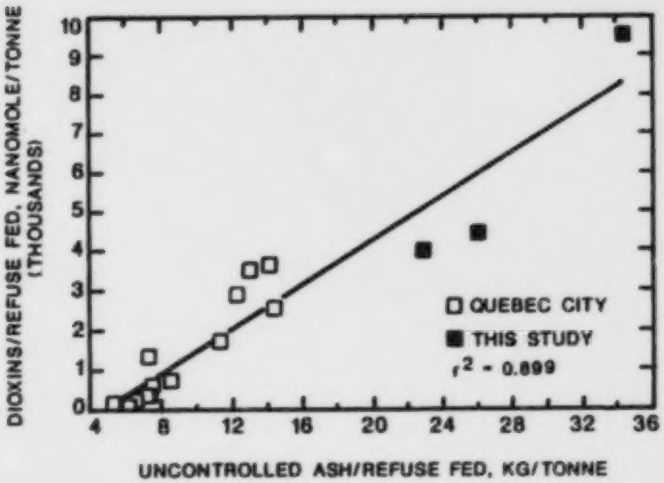


Figure 2. PCDD/PCDF Emissions vs. Uncontrolled Ash.

entrained from the furnace for the different operating conditions. A single correlation readily relates the higher RDF emissions from the emissions from the mass burn unit. Thus, RDF emissions of PCDD/PCDF may be higher than mass burn emissions due to the higher particulate carryover that occurs as a result of suspension injection and burning of RDF and the higher volumetric heat release rates of these units.

The data gathered under the GRI program at the RDF unit further indicates that PCDD/PCDF was largely formed within the furnace (see Figure 3); and down stream reactions appear to be less important for total emissions level. The importance of downstream reactions have been confirmed on a number of full scale tests for a number of different types of units. However, the test suggested that for this particular unit the impact of downstream reactions was to rearrange congener distributions rather than to impact total emissions level(6). Thus, even though total PCDD/PCDF did not increase, some downstream reactions did apparently take place. It is unclear whether the emissions would have increased in this particular unit if the entrained particles were held up in the system for longer times at 300°C, for example, if the system had a hot ESP. The data in other units suggest that PCDD/PCDF would have increased in this situation.

A hypothetical mechanism can be proposed that is consistent with the data from the test as well as a large volume of data from other types of municipal waste combustion systems. The mechanism relies on the escape of unburned material from the furnace. The most important material appears to be particulate matter. This material escapes from the waste combustion system due to the inadequate residence times, poor mixing, and low temperature pathways that can occur within furnaces burning the heterogeneous waste materials. These unburned materials can be more of a problem in an RDF combustion unit due to the mode of feeding, instabilities in the feed flow, and higher volumetric heat release rates. The PCDD/PCDF may actually form on surfaces of these particles due to localized pyrolysis zones. The data suggest that large particles may be most important.

The gas phase hydrocarbons that result from the partial decomposition of waste are another important component in the dioxin formation chain. These hydrocarbons may be converted to PCDD/PCDF within the furnace or downstream if the catalytic fly ash are held at 300°C for sufficient time. The gas phase precursors such as Polychlorinated phenols (PCP) or the resulting PCDD/PCDF, escape the furnace due to cold temperature pathways and unmixed conditions. If they are mixed at sufficient temperature, these organics will be destroyed almost instantaneously within the furnace(12).

The hypothetical mechanism described above has not been completely validated. However, it can be concluded that the destruction of PCDD/PCDF and precursors within the furnace will reduce emissions of PCDD/PCDF. The precursors are both other chlorinated dioxin and furans and gas phase hydrocarbons such as PCP's as well as unburned particle matter. Hence, a workable combustion control strategy would be to limit the emissions of these unburned species by establishing conditions in the furnace that encourage complete burnout of all combustibles. If the precursors escape the furnace,

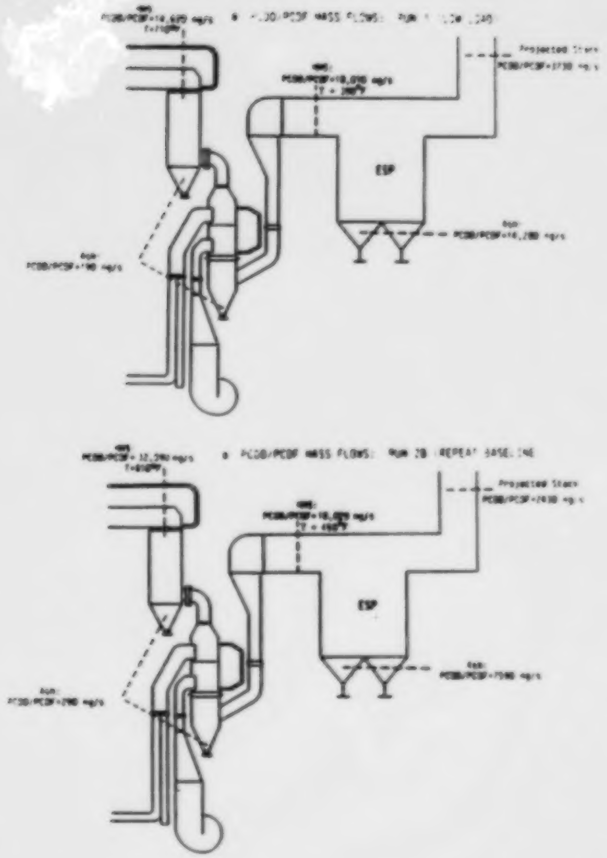

PCDD/PCDF MASS FLOWS: RUN 1 (LOW LOAD)
PCDD/PCDF=14,600 ng/s
Projected Stack
PCDD/PCDF=3730 ng/s
ESP
Ash:
PCDD/PCDF=14,280 ng/s
Ash:
PCDD/PCDF=190 ng/s
PCDD/PCDF MASS FLOWS: RUN 2B (REPEAT BASELINE)
PCDD/PCDF=12,290 ng/s
Projected Stack
PCDD/PCDF=2430 ng/s
ESP
Ash:
PCDD/PCDF=7090 ng/s
Ash:
PCDD/PCDF=290 ng/s

Figure 3. Mass flow rates of PCDD/PCDF estimated from concentrations in RDF plant.

then the particle hold-up time at critical temperatures should be minimized. The final strategy, therefore, is to quench the flue gas, thus forcing the condensation of PCDD/PCDF onto fine particles and then to efficiently remove the fine particle matter.

THE DEVELOPMENT OF NATURAL GAS FIRING TECHNOLOGIES FOR CONTROL OF CHLORINATED DIOXIN FURANS

The evaluation of how PCDD/PCDF are formed and how they can be controlled via natural gas cofiring has been the subject of pilot scale tests conducted at the EER test facilities in El Toro, California. The pilot scale test facility used for these studies is shown in Figure 4. The 0.5 MMBtu/hr unit is designed to fire actual RDF samples obtained from the NSP Wilmarth Station. RDF is gravity fed from the feed bin through the pneumatic feeder line which is injected along with air into the vertical tower furnace (12 in. ID). This pneumatic RDF feeding system simulates the performance of the actual Wilmarth feeders in which the fuel is blown across the width of the furnace allowing combustion to occur both in the suspension phase and on the grate. A refractory grate has been installed at the bottom of the facility. The grate is perforated to allow the injection of undergrate air. The undergrate air flow rate is an important parameter to controlling both feed combustion and particle entrainment in the furnace gas. The grate does not travel during furnace operation but is rather removed and cleaned after each test run.

Natural gas burners are located both in the lower furnace near the grate and higher up in the unit in a location above the primary combustion zone. The function of the lower burner is to control the total heat input and temperature profile of the system. The upper burner has been designed to allow evaluation of cofiring concepts by having the flexibility in its heat input, furnace flow fuel coverage, and combustion air stoichiometry.

Down stream of the furnace is located the system convective pass. Convective water cooled coils are located in this region to simulate heat removal in the economizer air heater and other convective passages in the full scale unit. By adjusting the heat input from the waste feed and auxiliary natural gas burner and the heat removal by the convective pass, it is possible to simulate the residence time, temperature history present in a full scale combustor. Downstream of the convective pass is located a bag house for particle removal. Modified method five sampling is performed at the inlet to the bag house. This zone is just downstream of the dioxin formation temperature window of 470-660°F. Thus, it is not expected that any significant dioxin formation will occur down stream of the sampling equipment.

The tests conducted to date on the pilot scale unit have focused on establishing the conditions similar to full scale RDF combustion systems and examining control options using natural gas cofiring. The PCDD/PCDF emissions for the pilot scale unit have been somewhat higher in these early tests than baseline operations on full scale units (4,000 ng/dscm as compared to 1500 ng/dscm). Several tests have been conducted including tests with different levels of natural gas cofiring and tests above and below the critical temperature. The firing of as little as 10% natural gas can result in over

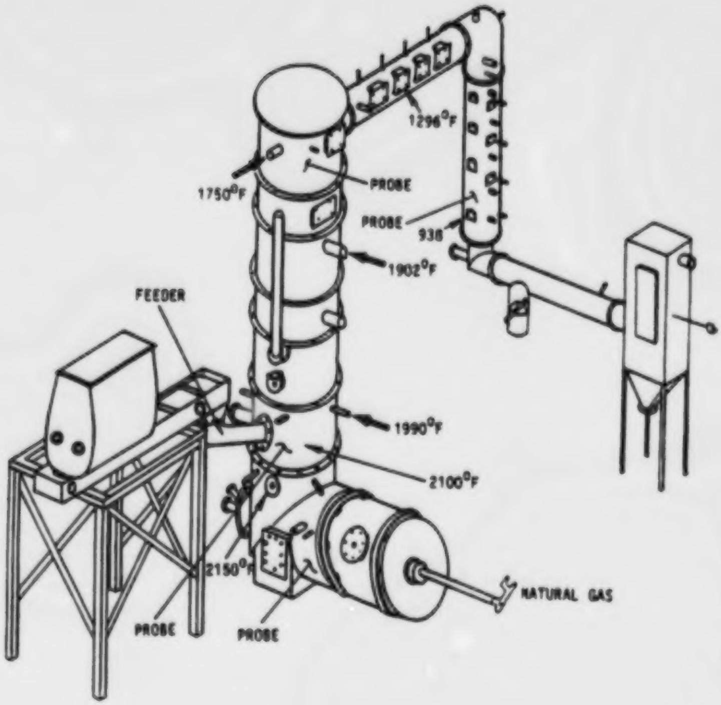

1296°F
1750°F
PROBE
PROBE
938
1902°F
FEEDER
1990°F
2100°F
2150°F
NATURAL GAS
PROBE
PROBE

Figure 4. Solid fuel pilot scale test facility at EER El Toro test site.

50% reduction of PCDD/PCDF; larger percentage firing rates and optimum firing of natural gas should allow over 90% control. The congener distributions achieved in the pilot scale facility are consistent with the full scale data showing a predominance of lower chlorinated furans. Ongoing studies will attempt to optimize the control of PCDD/PCDF with minimal gas cofiring.

GAS COFIRING DESIGNS FOR PCDD/PCDF CONTROL

The studies on the characterization of the full scale RDF unit and at the EER pilot scale facility have lead to the conclusion that PCDD/PCDF can be controlled via the retrofit of natural gas cofiring systems. The conceptual design of the retrofit system addresses the major routes of PCDD/PCDF formation and escape. These routes were identified by the characterization of the full scale RDF unit and an examination of formation escape mechanisms that have been identified through the extensive literature as well as a performance test carried out at full scale and the pilot scale unit. PCDD/PCDF formation mechanisms considered to be important include the following:

- Low temperature pathways which result as either spatial or temporal variations of the heat release within the furnace. These provide pathways by which PCDD/PCDF or precursors of PCDD/PCDF escape from the furnace. For RDF systems, these pathways likely result from feed rate and composition variations. Fluctuations in temperatures as large as 400°F can occur over the furnace cross section in relatively short time periods at any location within the furnace.

- Excessive amounts of unburned particles of RDF are entrained in the furnace flow and escape complete combustion due the short residence time and low temperature pathways. Some evidence suggests that these relatively large particles can be directly involved in PCDD/PCDF formation processes; perhaps, due to localized pyrolysis reactions around the particle.

- Downstream formation reactions which convert the precursors that escape the furnace to form PCDD/PCDF.

The evidence from the GRI field tests clearly indicates that downstream reactions occurring near 300°C did take place which rearranged homologue concentrations. Other data sources can be used to demonstrate that PCDD/PCDF increases if particles are held at 300°C such as with a hot ESP.

The natural gas cofiring scheme under development is attempting to address these mechanisms. Technical improvement criteria for the natural gas cofiring scheme suitable to control the PCDD/PCDF levels down to the goals of less than 100 ng/dscm are summarized in Table 2. These same criteria should be applicable to the control of CO. This table shows the PCDD/PCDF formation escape mechanisms being addressed, the improvement criteria, the current basis for the criteria and remaining issues that must be addressed.

TABLE 2. TECHNICAL IMPROVEMENT CRITERIA FOR A NATURAL GAS COFIRED SYSTEM FOR CONTROL OF PCDD/PCDF TO 100 ng/Nm^3

PCDD/PCDF SOURCE	IMPROVEMENT CRITERIA REQUIRED TO LOWER PCDD/PCDF TO GOAL OF 100 ng/Nm^3	HOW CRITERIA WAS ESTABLISHED	ISSUES REMAINING THAT MUST BE ADDRESSED
1. Low Temperature	• Reduce upper furnace temperature variations across cross section to ± 75°F	• Comparable to mass burn units	• Necessary damping level
2. Temperature Variation	• Reduce temperature fluctuations in upper furnace from 400°F in ten minutes to 150°F in ten minutes	• Comparable to mass burn units	• Necessary damping level
3. Precursor Destruction	• Subject 90% of furnace flow to flame zone environment	• Afterburner analogy • Kinetic rates enhanced by flame radicals	• Define whether this is a necessary requirement
4. Particle Carryover	• Reduce unburned particle carryover to 0.7 gr/dscf	• Particulate matter vs. PCDD/PCDF correlation developed from Wilmarth & Quebec City to correspond to 100 ng/Nm^3	• Techniques suitable to lower particulate entrainment
5. Particle Burnout	• Reduce entrained particle	• Correlations from Quebec City	• Define whether this is a necessary requirement
6. Particle Holdup	• Do not operate ESP in temperature window of 250-350°C (480-662°F)	• Fundamental data showing particle gas interactions in temperature window	• Must define if this is necessary requirement if other combustion improvement are made

Several design options can be proposed for natural gas cofiring schemes which will address these improvement criteria. In summary, whatever natural gas cofiring system that is employed must accomplish the following items:

- Provide uniform heat release across the furnace cross section above the last air injection point to damp fluctuations and temperature.

- Entrain furnace gas efficiently into the active natural gas flame zone.

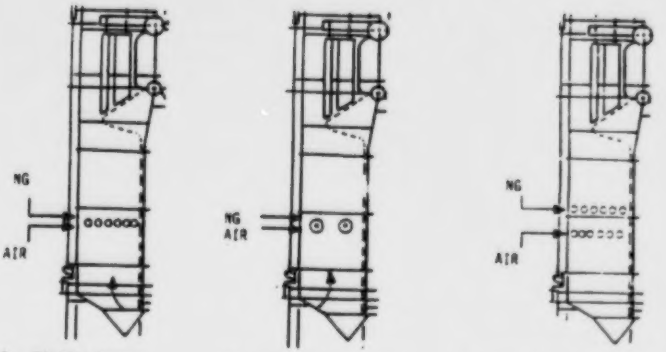
- Reduce particle entrainment from the lower furnace via manipulation of the air distribution (i.e. reduce underfire air and air injected below the RDF feeder. Provide as long a residence time as possible after the natural gas injection point for precursor burnout.

- Drop ESP operating temperatures below the critical temperature window.

Currently, five separate natural gas cofiring designs are being evaluated relative to the improvement criteria discussed above. These design options are summarized in Figure 5.

In the first two options, natural gas is added with combustion air and a stabilized burner arrangement. The first option involves mini gas burners which generate stable flames. Previous studies have indicated that 12 burners (6 on two opposing walls) will provide good coverage of the furnace. The second option would be similar in design but would employ larger burners that would impart significant mixing momentum. For both of these options, the underfire air will be lowered to reduce particle entrainment. The advantage of the first option is that the numerous burners will provide very good coverage of the furnace flow and can sometimes be arranged to not require pressure part modifications. In one simplified approach, the natural gas burners could be substituted for the last row of overfire air injectors. The second option allows the use of less hardware, but will require pressure part modifications (tube bending). The larger burners could provide as good of coverage but would provide a large momentum source for mixing.

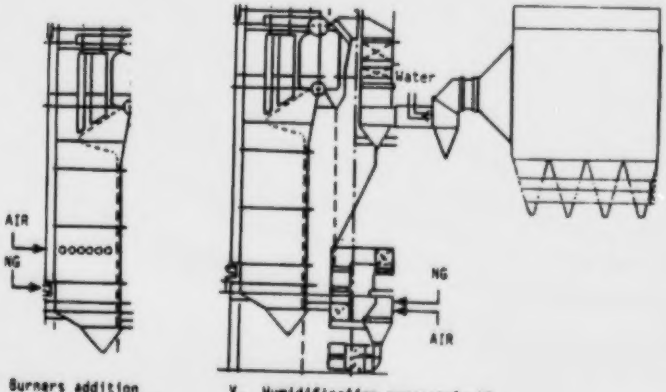
The third option is the separate injection of natural gas and combustion air in a reburning type of configuration. Reburning is currently being demonstrated by EER under support from GRI/DOE/Illinois for control of NO_x in coal fired utility boilers(13) and is being investigated at pilot scale for MWCs by GRI. The underfire air would be reduced as far as possible to lower particle entrainment and move to the overfire air injection location. The natural gas might require addition of recirculated flue gas in order to have sufficient momentum to mix with the furnace flow or high pressure injection can be used. This technique delays the mixing of the gas with combustion air and allows it to mix prior to oxidation. In this manner, the full combustion volume becomes involved in flames which can enhance radical attack on chlorinated organics. This is analogous to the use of natural gas reburning reactions where radical attack on NO_x leads to NO_x reduction. The main disadvantage of this technique are the added complexity, especially if FGR is


NG
AIR
NG
AIR
NG
AIR

I. Minigas burner reduced UFA

II. Large gas burners reduced UFA

III. Separate NG & Air injectors reduced UFA


AIR
NG
Water
NG
AIR

VI. Burners addition at RDF feed reduced UFA

V. Humidification, separately NG fired air heater

Figure 5. Natural gas cofiring options for RDF plants for control of PCDD/PCDF.

required and safety since non stabilized flames are employed. However reburning is a technology that has been applied successfully to full scale furnaces indicating that these restrictions can be overcome.

The fourth option involves the addition of natural gas to the RDF injection location. This can be accomplished via the burners in the lower furnace or by modification of the air spout distributors. This has the advantage of putting natural gas directly in contact with the RDF. However, it has the disadvantage of modifying the basic RDF combustion process.

None of the first four options deals with downstream processes. The fifth option addresses this directly via the addition of water spray humidifier to lower the ESP temperature. A separately fired natural gas air heater is then used instead of the flue gas air heater. Disadvantages of this technique is that it only removes PCDD/PCDF, will require ESP modifications and cannot be used for start-up or load following.

FULL SCALE EVALUATION OF GAS COFIRING TECHNOLOGIES AND RDF COMBUSTORS

The Gas Research Institute is currently in negotiation with a owner of a full scale RDF combustion system for evaluation of the gas cofiring technology for control of PCDD/PCDF, start-up and load following. This demonstration full scale evaluation will involve the retrofit of a gas cofiring technologies appropriate for the specific system. The study will include:

1. Design of gas cofiring to meet the specific needs of the host RDF unit which is representative of gas RDF cofiring applications.
2. Installing the gas cofiring system
3. Testing the host unit under baseline conditions and with gas cofiring
4. Analyzing the test data to clearly demonstrate the performance goals are met for the host unit.
5. Utilizing the test data to define design approaches for gas firing applications to other types of MSW units and to project performance improvements.
6. Provide a mechanism of transferring the technology to owner/operators of other MSW systems.

This study is planned for the next 18 months.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and, therefore, the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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THERMAL DESTRUCTION OF PCDD/PCDF COMPOUNDS BY 3R PRODUCT RECYCLING INTO THE SOLID WASTE INCINERATION FURNACE

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ABSTRACT

Recycling of 3R products into the incinerator of a municipal solid waste incineration plant for thermal destruction of PCDD/PCDF compounds has been demonstrated on an industrial scale and supported by an extensive program of measurements.

An essential requirement for this recycling was an upstream product conditioning step in which dimensionally stable pellets were compacted in a mixer-extruder. A tracer which could be activated has been added to the product mixture for indicating potential releases of pellet dust into the raw gas.

A large number of pellets were sampled from the bottom ash stream as a way of monitoring the experiments. In a parallel step, fly ash samples were separated by an isokinetic sampling method of the raw gas duct downstream of the boiler.

Chemical-analytical examinations show that the PCDD/PCDF compounds are destroyed almost completely after thermal treatment in the solid waste incinerator. Significant dust releases can be excluded as a consequence of the compaction technique chosen. No elevated PCDD concentration can be detected in the raw gas.

INTRODUCTION

In the treatment of filter dusts from municipal solid waste incineration plants by the 3R process (1,2), the residual solid remaining after acidic heavy metal extraction is to be recycled into the incineration for thermal decomposition of the PCDD/PCDF compounds (3). However, a conditioning step must be carried out first, in which the solid, which is present as a filter cake, is compacted into a dimensionally stable product resistant to the mechanical and thermal stresses encountered in the refuse incineration plant. There must be no quantitative release of secondary process dust; instead, the product is to be discharged together with the bottom ash from the refuse incineration, and the recycling is to render the quality of the product identical to that of the bottom ash.

This paper describes two recycling experiments performed on two different days. They were carried out at the Oberhausen municipal solid waste incineration plant; the 3R product recycled into the incinerator quantitatively corresponded to the fly ash released in that incinerator. This paper is organized in two sections: The technical process part describing product conditioning, recycling, and the sampling techniques, and a chemical-analytical part dealing with the quantitative determination of PCDD/PCDF inventories and potential dust releases into the flue gas.

ASPECTS OF PROCESS TECHNOLOGY

COMPACTION OF 3R INTERMEDIATE PRODUCTS

In the concept of the 3R process, whose flowsheet is shown in Figure 1, the solid/liquid separation step produces as a residue a filter cake with a residual moisture of approx. 30 %. In this form, this intermediate product cannot be recycled because it would largely be released again as additional process dust in the incinerator and carried away with the flue gas. The recycling problem will be solved as soon as the filter cake can be compacted into dimensionally stable pellets by some machine.

On the basis of tentative laboratory-scale tests, a method of compaction in a mixer-extruder was developed in cooperation with the Werner + Pfleiderer company, Stuttgart, a machine manufacturer. In a continuous extrusion process, the filter cake with a defined moisture together with a binder is formed into a cylindrical product (pellet size: approx. 40 mm diameter, approx. 80-120 mm length).

In order to be able to trace quantitatively in the flue gas system any releases of pellet dust resulting from the thermal treatment in the refuse incinerator, a tracer was added in the compaction step, which can be activated and then detected as a component of the flue dust.

The tracer chosen was tantalum in the chemical form of Ta_2O_5, which was added at a concentration of 1000 ppm. At a variance of less than ± 10 %, the homogeneity in the pellets is sufficient for the recycling experiments.

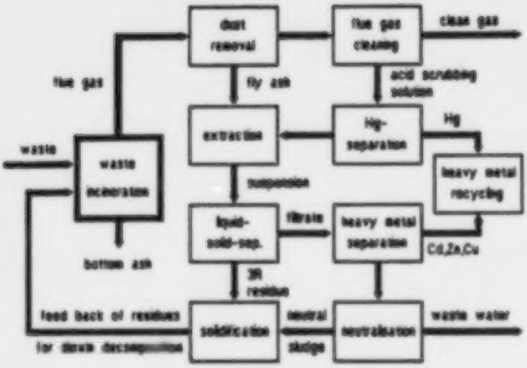
dust removal
flue gas cleaning
clean gas
flue gas
fly ash
acid scrubbing solution
extraction
Hg-separation
Hg
waste
waste incinerator
heavy metal recycling
suspension
liquid-solid-sep.
filtrate
heavy metal separation
Cd,Zn,Cu
bottom ash
3R residue
feed back of residues
for dioxin decomposition
solidification
neutral
sludge
neutralisation
waste water

Figure 1. 3R process flowsheet

The tracer was selected in light of the following criteria:

1. The tracer contents in the flue dust matrix is either zero or only negligibly low.

2. The chemical compound of the tracer is not volatile in the temperature field of the incinerator.

3. The tracer can be determined with a high sensitivity of detection, which is done in this case by neutron activation analysis with a limit of detection of approx. 0.5 µg/g.

RECYCLING EXPERIMENTS

The two experiments of recycling compacted 3R residues into the incinerator in order to decompose PCDD/PCDF compounds were run at the Oberhausen municipal solid waste incineration plant. Figure 2 shows a schematic diagram of the incinerator designed for a nominal throughput of 20 t/h of municipal solid waste.

To simulate continuous recycling, 100 kg pellets, packed in 10 kg containers, were fed into the refuse hopper every 10 minutes. The dosage time in each case was 2.5 h. At an overall retention period of approx. 2 h between the feed and and the discharge from the grate, the pellets are exposed to a thermal treatment on the roller grate for about 45 minutes.

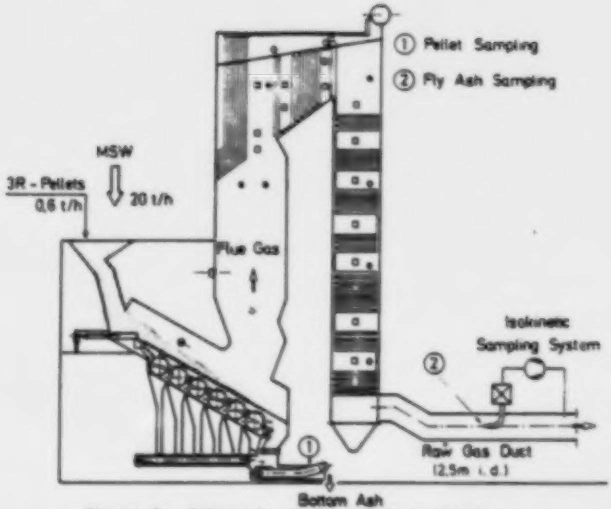

Figure 2. Schematic of the MSW incinerator

Positions [1] and [2] in Figure 2 characterize the sampling points. At position [1], pellets were continuously removed from the bottom ash stream by manual sampling techniques. At position [2] in the raw gas duct between the boiler and the electrostatic precipitator, an isokinetic sampling system was installed for separating flue dust and condensing and adsorbing gaseous PCDD/PCDF compounds by extracting a bleed stream. As will be outlined below, the flue dusts were examined both for organic pollutant compounds and for traces of tantalum.

Prior to the recycling experiments, the isokinetic sampling system (4) shown schematically in Figure 3 was used for extensive network measurements of local and temporary flue dust loads of the gaseous effluent stream. Because of the horizontal installation of the raw gas duct, with two flow baffles downstream of the boiler, considerable concentration gradients had to be assumed to exist across the section of the raw gas duct. These network measurements were to be used in developing a sampling strategy ensuring that the sampling material extracted and separated is representative with respect to the dust concentration and the particle size distribution. The results of these measurements as shown in Figure 4 indicate that, because of the proven extremely asymmetrical dust distribution, a scanning technique was advisable. For this purpose, the extraction probe had to be put into seven different extraction positions situated on a circle in the course of one measurement cycle.

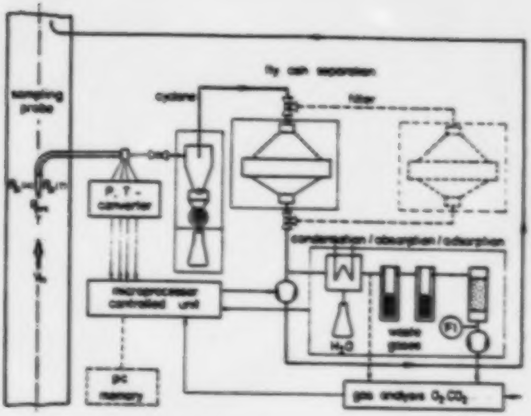
sampling probe
fly ash separation
cyclone
filter
P, T - converter
microprocessor controlled unit
pc memory
condensation / absorption / adsorption
H_2O
waste gases
FI
gas analysis O_2 CO_2

Figure 3. Schematic of the isokinetic sampling system

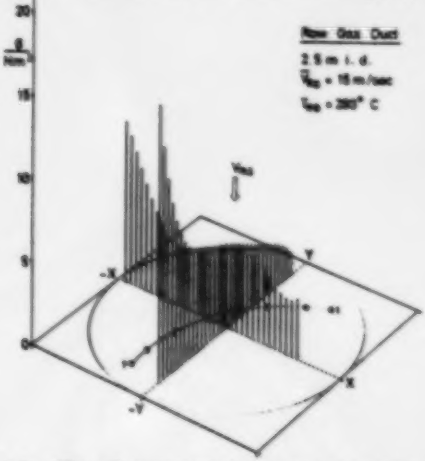

Figure 4. Fly ash concentration across the cross section of the raw gas duct

For each test day, an average dust loading of 6.5 ± 2.4 g/Nm^3 with a median particle size of 60 µm was determined from the scans. In general, over 90 % of the flue dust volume separated by the isokinetic sampling method could be removed in the measurement cyclone.

CHEMICAL-ANALYTICAL FINDINGS

RESULTS OF PELLET ANALYSES

On both test days, the first pellets appeared on the bottom ash discharge belt two hours after the onset of the feeding process. Most of the pellets were of a greyish brown color, showing clear traces of the thermal impact encountered on the grate. Some pellets had not changed shape in any way, while others had been fragmented into shorter pieces. In general, no major wear and tear was recognizable. This already allows to draw the conclusion that hardly any dust has been released from the 3R material on the grate.

Some pellets appeared on the bottom ash discharge belt more or less unchanged in color. These must have been pellets which had fallen between two rollers at some point on the roller grate as a result of insufficient sealing, and thus had not passed through the high-temperature incineration zone as planned. The substances falling through the grate are combined with the discharge from the grate at the Oberhausen municipal solid waste incineration plant.

At intervals of 30 minutes some 6 - 10 pellets were taken from the bottom ash discharge belt at random and used for archive purposes. Twelve samples were selected a day to be analyzed for PCDD and PCDF. As a reference, the average analysis of three green pellets was used, for which the following analytical data had been found:

$$C_0(\text{PCDD}) = 400 \pm 80 \text{ ng/g},$$
$$C_0(\text{PCDF}) = 240 \pm 40 \text{ ng/g}.$$

Figures 5 and 6 show the PCDD and PCDF residues in the pellets analyzed in % of the initial concentration C_0, plotted versus the discharge time for both test days. All PCDD residues found on the first day of the test (October 18, 1988) were below 0.5 %, which corresponds to a concentration of 2 ng/g. Ten out of the twelve analytical results even fell below 0.15 % or 0.6 ng/g. The analytical limit of detection was 0.03 ng/g. The dioxin results are striking, inasmuch as none of the samples contained tetra-, penta- or hexachloro compounds.

The result is slightly less favorable for PCDF, which are known to be thermally more stable than PCDD. As is seen in Figure 5, two pellets had peak levels of 3 and 1.5 %, respectively, containing all possible degrees of chlorination, especially also up to 5 ng/g of T4CDF. Comparisons with the other results, which contained no P5CDF, H6CDF, and H7CDF in six cases and only small quantities of these compounds in the other cases, indicates

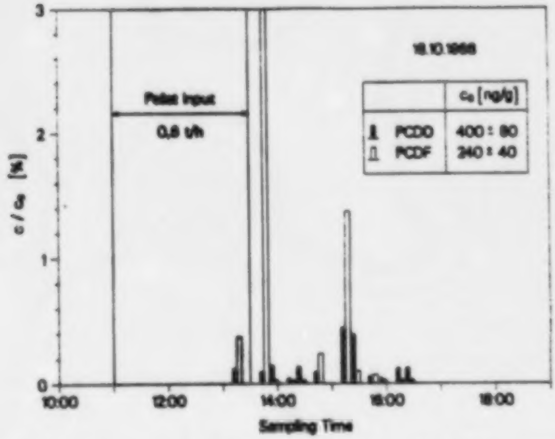

Figure 5. Relative PCDD/PCDF concentration in pellets vs. sampling time

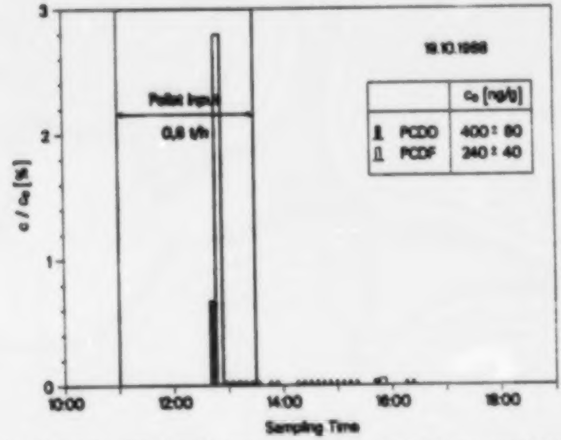

Figure 6. Relative PCDD/PCDF concentration in pellets vs. sampling time

conclusively that these two samples with their outlying results must have been those which had fallen through the grate.

On October 19, 1988, the overall result was even better than on the previous day, as can be seen from Figure 6. Positive detection was possible only in two samples, while the other ten samples did not indicate any traces of PCDD or PCDF within the detection limit of 0.03 ng/g. The first sample, taken at approx. 12.45 p.m., was characterized by high concentrations of PCDD (2.7 ng/g) and PCDF (6.7 ng/g). While only H7CDD and O8CDD were found for the dioxins, all chlorination steps were found among the furanes, especially also T4CDF. As with the two other extreme values found on the first day, it may be concluded from these findings that also this pellet cannot have passed through the whole incinerator, but must have dropped through the grate.

In the second pellet with a positive finding, the levels of 0.15 ng/g for O8PCDD and 0.13 ng/g for H7CDF were found to be only slightly above the limit of detection.

If one looks at the total number of analyses on both days, one must classify at last three pellets with extreme PCDF levels under the heading of pellets fallen through the grate. Consequently, the conclusion to be drawn also from these experiments is the requirement not to combine them with the bottom ash that falls through the grate, but rather return them to the feed hopper of the municipal solid waste incinerator.

In the other 21 out of 24 samples, maximum residues of less than 0.4 % of PCDD or PCDF were found. In ten cases, no PCDD was found at all, and only in three cases there were small quantities of H7PCDD. The complete absence of H6CDD, P5CDD, T4CDD, and also of T4CDF means that all isomers of high toxicity are destroyed completely and that later assays of 2,3,7,8-T4CDD equivalents will give rise to extremely low levels.

The large-scale experiment thus confirms the conclusions drawn from earlier experiments in laboratory-scale furnaces and in the TAMARA test facility for municipal solid waste incineration at the Karlsruhe Nuclear Research Center, that namely 3R products contain practically no more PCDD and PCDF after having been recycled through the incinerator (5,6).

RESULTS OF RAW GAS SAMPLING

As outlined above, the raw gas stream was to be sampled especially in order to find answers to the following two questions:

- Is there any additional dust release during the recycling experiment, which would also mean increased dust loads in the raw gas?

- Will there be increased PCDD and PCDF levels in the raw gas because PCDD and PCDF from the recycled 3R products is volatilized and perhaps not destroyed completely in the incinerator and the afterburner chamber, respectively?

Figures 7 and 8 are plots of the concentrations (in g/Nm^3) measured of fly ash and of PCDD and PCDF (in ng/Nm^3) on the same time scale for the two test days. Because of the possible retention times of the waste in the feed hopper and on the incinerator grate it may be assumed that 3R material was on the incinerator grate between approximately 12 noon and 3.30 p.m. On the first day, three feed (preliminary period) and four discharge (final period) samples of the raw gas were taken, while three feed and three discharge gas samples were taken in the second day. During pellet operation, the samples were taken at an average of 30-minute intervals.

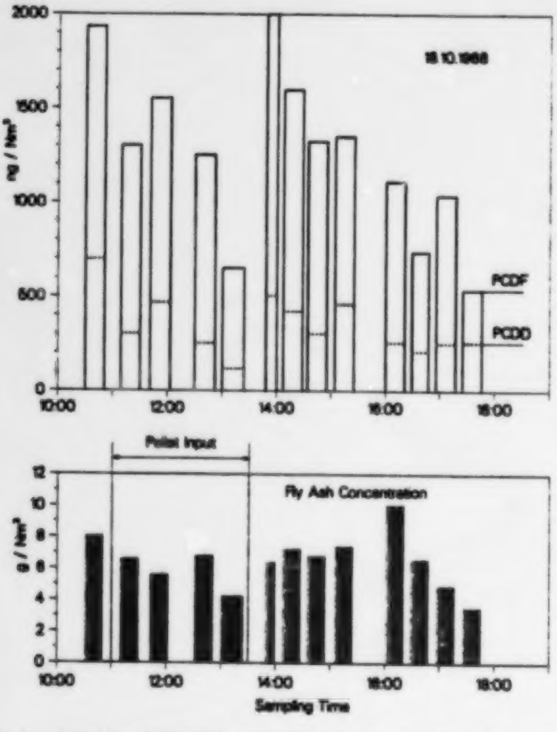

Figure 7. Specific PCDD/PCDF concentration in the fly ash samples and fly ash concentration in the raw gas vs. sampling time

The two diagrams show that both the dust concentrations and the PCDD and PCDF contents vary greatly. On the basis of these plots, no significant change within the period of the recycling experiment can be seen for any measured quantity.

A more precise analysis of what is going on can be obtained by comparing the mean values found in each sampling phase, which are listed in Table 1.

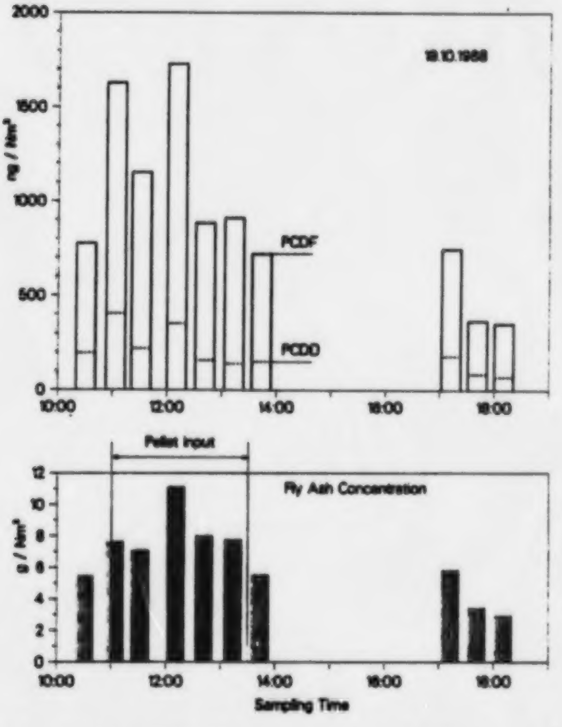

Figure 8. Specific PCDD/PCDF concentration in the fly ash samples and fly ash concentration in the raw gas vs. sampling time

In the third column of this table, we find the fly ash concentrations and their variances. On the first test days, they show no striking features during 3R product recycling; the level found is precisely between those of the feed and the discharge samples. On the second day, however, on which only four samples were taken during recycling operation, a significant increase in the fly ash contents of the raw gas is found in this time span of 8.1 g/Nm^3 over 6.8 g/Nm^3 in the feed and only 4.1 g/Nm^3 in the discharge samples.

TABLE 1. RESULTS OF THE CHEMICAL-ANALYTICAL INVESTIGATIONS

Date (1988)	Sampling Phases	Fly Ash g/Nm^3	Tantal µg/g	PCDD ng/Nm^3	T4CDD ng/Nm^3	PCDF ng/Nm^3	T4CDF ng/Nm^3
18.10.	(3) Preliminary Period	6.8 ± 1.2	3.7 ± 0.5	480 ± 200	28 ± 6	1800 ± 320	350 ± 40
	(8) Pellet Experiment	6.6 ± 1.2	3.7 ± 0.7	340 ± 150	18 ± 7	1370 ± 460	330 ± 100
	(4) Final Period	6.3 ± 2.8	-	246 ± 26	12 ± 8	880 ± 230	190 ± 70
19.10.	(3) Preliminary Period	6.8 ± 1.1	3.7 ± 0.6	270 ± 120	18 ± 14	1200 ± 110	320 ± 110
	(4) Pellet Experiment	8.1 ± 2.3	3.3 ± 0.8	200 ± 100	14 ± 10	1080 ± 460	330 ± 160
	(3) Final Period	4.1 ± 1.6	-	110 ± 60	10 ± 12	460 ± 230	120 ± 60

As described above, the compacted 3R products had been labelled with tantalum (1000 ppm) for reliable detection of the fly ash fractions originating from these materials. The dust samples extracted therefore were subjected to a neutron activation analysis of their tantalum contents. The result of this analysis, which is listed in column 4 of Table 1, shows that the mean level of 3.6 ± 0.8 ppm implies identical concentrations of tantalum in the fly ash. As 1000 ppm of tantalum is contained in the feed material, releases on the order of 1% should result in a 10 ppm increase in the tantalum concentration in the flue dusts. If one assumes an accuracy of ± 1 ppm for the analyses of fly ashes, it follows that, within the limits of measuring accuracy, no increase in the natural tantalum content, and thus no dust release from the 3R products, can be detected. The increase to 8.1 g/Nm^3 found in the fly ash content therefore cannot be due to material releases from the 3R product.

The last four columns of Table 1 summarize the analytical data of PCDD and PCDF and the respective tetrachloro-compounds to be observed in particular under toxic aspects. It should be added that the gas phase fraction was always clearly below 1 %. Also the curve of the mean values does not indicate any significant increase in concentration during the recycling experiment. Apart from T4CDF, all the other mean values continuously decline throughout the day.

An estimate shows that a release of approx. 10 % of the inventory of 3R materials would have to raise the PCDD concentration by some 200 ng/Nm^3 and the PCDF concentration by approx. 120 ng/Nm^3 in the raw gas. The measured data and their variances show for PCDD that the real amount added is clearly below these 10 % and one therefore can assume the PCDD feed to have been destroyed almost completely.

Because of the larger variance of the mean values rough estimates of PCDF releases are more difficult to make. The concentration decline observed in the course of a day seems to be less pronounced with PCDF, especially T4CDF, than with PCDD. This may indicate that, because of greater thermal stability, a certain fraction of volatilized, undestroyed PCDF must be taken into account, meaning that the "internal dioxin circuit" postulated on an earlier occasion is likely to exist at least for PCDF. However, in the light of the measured results available, no quantification is possible.

As far as more than 99 % of the PCDD and PCDF found in the raw gas was bound to fly ash, it may be assumed for a municipal solid waste incineration plant with good dust separation capability that recycling of 3R products to the incinerator grate may lead to a slightly higher level, especially of PCDF, in the raw gas, but should not have any consequences on the emission side. This should be supported by further experimental measurements, which have already been planned.

SUMMARY AND EVALUATION

Recycling 3R products into the incinerator of a municipal solid waste incineration plant for thermal decomposition of PCDD/PCDF compounds has been demonstrated on an industrial scale and supported by an extensive program of measurements.

This recycling required upstream conditioning, in which dimensionally stable pellets were compacted in a mixer-extruder.

For subsequent checking of the experiments, several pellets were sampled from the bottom ash stream after having passed through the incinerator, and a sufficient number of dust samples were taken from the raw gas duct downstream of the boiler by isokinetic bleeding extraction.

The results of the recycling experiment can be summarized as follows:

- The contents of PCDD and PCDF in the 3R products are reliably destroyed down to negligible residual concentrations when passing through the incinerator of a municipal solid waste incineration plant. For industrial application, material dropped through the grate must not be combined with the discharge from the grate but must be returned to the refuse feed hopper.

- As a consequence of the method of compaction chosen for the 3R filter cake, no additional dust releases were observed in the raw gas duct.

- Within the limits of measuring accuracy, no increased PCDD load due to the pellets can be detected in the raw gas. Small amounts of PCDF, which have been released and perhaps not been destroyed completely on their way through the incinerator, cannot be excluded entirely. However, they will hardly give rise to elevated emission concentrations.

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CURRENT ISSUES AND PROBLEMS IN INTEGRATED SOLID WASTE MANAGEMENT

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Fairfax, VA 22031

CURRENT ISSUES AND PROBLEMS IN INTEGRATED SOLID WASTE MANAGEMENT

With decreasing landfill capacity, rising disposal costs, and increasing environmental concerns, decision makers at the state, county, and local levels are faced with difficult decisions in solid waste management planning. Alternatives such as recycling, waste-to-energy recovery, and composting are beginning to play a more prominent role in solid waste management. Decision makers are currently combining these alternatives into integrated solid waste management plans. However, there are several problems with an integrated solid waste management approach including economic considerations, public acceptance, and regulatory concerns. As a means of assisting these decision makers, we have reviewed the current issues and problems in integrated solid waste management for EPA and are currently preparing a Guide that will help decision makers design and implement integrated solid waste management plans. This paper presents those current issues and problems we have encountered and briefly discusses what will be found in the decision makers guide.

1. WHAT IS INTEGRATED SOLID WASTE MANAGEMENT?

Integrated solid waste management is the ability to reduce the amount of waste to be landfilled by implementing and integrating several waste management options, such as waste reduction, recycling, composting, and energy recovery. The benefits derived from integrating these waste management options include extended landfill lifetime capacity, lower disposal costs, and lower environmental risks.

A successful integrated solid waste management plan generally includes the following steps:

- Reduce the amount of waste generated;
- Recycle everything possible;
- Recover energy from the remaining combustible materials; and
- Bury the remaining non-combustibles and ash if they cannot be used.

Several communities do not have a fully integrated solid waste management program but do use one or more of the management options. For example, a town may only have a landfill and a recycling program, or a waste-to-energy recovery system and landfill.

2. WHAT TRIGGERS INTEGRATED WASTE MANAGEMENT?

Traditionally, landfills have been used to manage and dispose of waste, though incinerators have become more popular over the past ten years. However, problems such as a lack of capacity for landfills or hazardous residuals (e.g., emissions and/or ash) from incinerators have lead decision makers towards a more integrated approach of waste management including an increased reliance on recycling and composting.

2.1 Capacity Crisis

In most cases, the lack of landfill capacity triggers consideration of integrated waste management. For example, in Mecklinberg County, North Carolina, the city began developing a solid waste plan in 1983 when it was determined that the city and county landfills would be filled to capacity by about 1986. The governing body of Mecklinberg is the Board of County Commissioners who have ultimate approval over solid waste management plans. Eighty-five percent of Mecklinberg County is made up of the city of Charlotte, with the county's population totaling 465,000 people. In 1984, city and county solid waste officials developed options for managing waste within the County. The objectives and goals were discussed and plans were drawn up in conjunction with a citizens' advisory board that was designed to provide public input into the process. A variety of different options were explored from the beginning, including landfilling, incineration, recycling, and source reduction. Important factors in examining these options included economics, environmental concerns, and political feasibility. The plan ultimately developed by the county established a hierarchy for waste management consisting of recycling, incineration, and landfilling. The plan was approved by the Charlotte city council and the County Board of Commissioners. Because of early and constant public involvement, and cooperation between the city and county governments, the county did not have serious problems implementing the recycling program or siting the incinerator. Today, the county is constructing the incinerator, undertaking recycling, and continuing to operate a landfill. [1]

Fairfax County, Virginia began planning for alternatives to its Lorton landfill in 1979 because of concerns about future capacity. The Fairfax County Board of Supervisors is ultimately responsible for approving county solid waste plans prepared by the county solid waste staff. Initially, Fairfax focused on incineration as the only alternative to landfilling. The staff began developing potential sites for the incinerator and formed a citizens' advisory committee to participate in the review process for the incinerator. A variety of economic and environmental studies were performed for the proposed incinerator. As planning for the incinerator progressed, the citizens advisory council began pushing for recycling in conjunction with the incinerator. Eventually, as public opposition to the incinerator grew, the county agreed to undertake recycling with the hope that this would assist in approving the incinerator. This ultimately proved successful as the incinerator was recently approved. Today Fairfax disposes of its waste in the Lorton landfill, conducts limited paper recycling, and will begin using its incinerator in the early 1990s. The county intends to increase its recycling and composting activities in future years. [2]

There are numerous communities and counties in the United States that, facing a capacity crisis, have opted to incinerate the majority of their waste. As an example, the City of Akron, Ohio constructed a waste-to-energy facility in 1979 designed to handle 900 tons per day of municipal solid waste because the city's landfill was reaching capacity. A host of economic and technical information was reviewed by decision makers prior to approving the plant. The project initially experienced a series of technical problems that were overcome through additional funding. Today the facility has been reconstructed and handles most of the city's solid waste. [3]

In some cases the lack of incineration capacity triggers recycling. The Recycling Subcommittee of the Alexandria/Arlington Waste-to-Energy Task Force is currently investigating ways of developing an organized commercial and residential recycling program that can be integrated with the operations of its existing waste-to-energy facility. This recycling effort was triggered by the fact that the waste-to-energy facility is currently operating at full capacity and cannot handle any growth in solid waste generation from Alexandria and Arlington. The investigation involves determining the volumes of recyclables generated versus quantities effectively captured by a recycling program, the heat/energy content trade-offs of materials removed from the waste stream, and the available markets for accumulated recyclables. The recycling effort is being encouraged by the Virginia General Assembly, which enacted in 1988 a law allowing localities to enact mandatory trash separation ordinances. [4]

2.2 Economic Concerns

Increases in landfill tipping fees can trigger the start of a recycling program. The San Jose City Council and the city solid waste department are responsible for solid waste management in San Jose. In 1981, San Jose's garbage collection contractor requested a rate increase of 88 percent from the City Council. Although the City Council reduced this to 42 percent, this spurred them to consider alternative means of waste disposal. First, the city sought competitors for its collection contract but found no other company would bid because the existing contractor owned the landfill. To make the collection contract more competitive, the council therefore decided to site another landfill within the city. In addition, in 1983 the council adopted a number of program goals to shift the emphasis of solid waste from disposal activities to recycling activities.

The siting of the new landfill ultimately proved successful, forcing the existing collection contractor to drastically reduce costs in order to maintain San Jose's collection contract. In addition, the siting of this landfill allowed the city solid waste bureau to design the tipping fees and collection contracts in such a way that they actively encouraged recycling. Gary Liss, Director of the Program, described the changes in San Jose's management scheme as tearing apart the previous disposal oriented program and gearing it towards recycling. The city council has set a goal of recycling 25 percent of its waste by 1990 and the city currently operates one of the largest recycling programs in the country.

Other reasons for increases in disposal costs include monopoly situations. In San Jose, until a second landfill was sited, the local garbage collection agency had a virtual monopoly on land disposal. Through this monopoly, the collection agency attempted to greatly increase disposal costs. [5]

2.3 Environmental Concerns

There are serious environmental concerns with solid waste management. Wastes may leach from landfills and contaminate local drinking water supplies, or dioxin and metal emissions may result from municipal waste incinerator operations. For example, in Byron, Illinois, an unpermitted landfill at a salvage yard contaminated nearby groundwater and drinking water with PCBs, volatile organics, metals, and cyanide. [6]

To lower the environmental risks associated with the disposal of municipal wastes, EPA has proposed in 40 CFR 258 new criteria for municipal solid waste landfills (MSWLFs). These proposed regulations affect:

- MSWLF design and operation -- The regulations would require that new landfills be designed with liner systems, leachate collection systems, and final cover systems to meet design goals at the point of compliance. It would also require MSWLF owners/operators to conduct activities related to groundwater protection (i.e., groundwater monitoring).
- MSWLF location -- The proposal would affect the siting of new landfills and the expansion of existing landfills.
- State/Federal relationship in Subtitle D management -- The proposal would require States to revise their Solid Waste Management Plans to comply with the new Subtitle D requirements.
- Subtitle D enforcement authority; and
- MSWLF financial requirements -- MSWLF owners/operators would be required to demonstrate financial assurance for the costs of conducting closure, post-closure care, and, if applicable, corrective action for known releases.

2.4 Political Concerns

Some politicians and decision makers are convinced that recycling will increase the safety and reduce the cost of solid waste management. By finding an alternative to land disposal, such potential problems as wastes leaching into ground water can be avoided, as well as lowering disposal costs. In Peterborough, New Hampshire, a mandatory recycling program was implemented as an alternative to land disposal by the town's elected officials. The recycling program is not highly profitable (the township earns $400/year from recycling). However, in terms of avoiding disposal costs, the township has saved $16,000. This has made recycling an attractive alternative to land disposal and has been an incentive for the town's elected officials to increase the township's recycling efforts. [7]

In some cases a recycling program may be instated because there is great public demand or public pressure. The recycling program in Fairfax County was implemented due to the general public's negative response to a proposed energy resource recovery facility. Implementing the recycling program was instrumental in obtaining the incinerator's permit.

2.5 Regulatory Concerns

State and local governments have enacted legislation to discourage the continued practice of land disposal and encourage recycling and other waste management options.

- The State of Minnesota has enacted a law that will prevent the disposal of unprocessed garbage by 1990.
- The Virginia General Assembly has passed a law allowing localities to enact mandatory trash separation ordinances.
- The Commonwealth of Pennsylvania has recently enacted legislation mandating that within several years, communities must meet a 25 percent recovery rate from municipal solid waste.

3. HOW TO HELP DECISION MAKERS DESIGN AND IMPLEMENT INTEGRATED SOLID WASTE MANAGEMENT PLANS?

3.1 The Two Volumes

Because of the potentially diverse audience for the guide, the guide will be organized into two separate volumes -- one to serve the needs of decision makers responsible for the broad policy decisions and the other to meet the needs of individuals responsible for the more technical aspects of solid waste management. The first volume will provide a broad overview of the various policy options available to a local community in collecting and disposing of its solid waste and provide a description of the financial steps that must be taken to implement these various policies. This volume will also describe the various initial types of studies that should be undertaken before a particular policy is chosen (such as a waste analysis, preliminary cost analysis, and an assessment of possible state regulations constraining local decision makers).

The second volume will provide a detailed technical description of each of the disposal options (e.g., recycling, incineration, landfilling, and composting) and cover the technical and economic aspects of collecting solid waste. This volume will serve the needs of those decision makers responsible for the technical aspects of solid waste management.

3.2 Step By Step Approach of Volume I

The first volume to the decision makers guide will be divided into six chapters.

Chapter One - Introduction

Chapter One of the decision makers guide is divided into four parts and will accomplish four tasks:

- Describe the solid waste crisis in the United States;
- Introduce the reader to the concept of integrated solid waste management;
- Outline the purpose of the primer; and
- Indicate the other parts of the guide as well as other aspects of USEPA's efforts to fill the gap in integrated solid waste management practices.

Chapter Two - Where Do You Stand: The Political and Institutional Context of Integrated Solid Waste Decision-Making

Chapter Two will discuss the political and institutional environment surrounding solid waste decision makers. It will highlight the factors that need to be considered when decision makers are identifying and selecting options to address their community's solid waste management problem.

Chapter Three - Waste Management in Your Community: Analyzing the Problem

Chapter Three will help decision makers begin to ask the right questions and provide guidelines on how to formulate answers to these questions. This chapter will help the user begin to analyze the characteristics of the waste stream and future waste disposal requirements, as well as analyze the present waste management system. It will ask questions, present issues to be addressed, and provide a framework for the decision maker's preliminary approach to the problem. This chapter establishes an agenda of discrete activities and studies required to move toward an integrated approach to waste management.

Chapter Four - Reviewing Program Approaches, Technical Options, and Important Issues

Chapter Four will present brief introductions to a variety of program approaches and technical options for implementing an integrated waste management program. This chapter will also provide a brief discussion of important issues related to integrated waste management programs. Key areas for intensive investigation will be identified and reviewed.

Chapter Five - Establishing Short-Term and Long-Term Objectives

Chapter Five will help decision makers synthesize their thoughts from Chapters Two, Three, and Four. This Chapter will focus the decision maker's assessment of the political and institutional landscape, waste characterization, and current waste management system into a coherent, preliminary image of what the possible directions and alternatives are, what the constraints on the management approach are, and what appropriate end results should be. This chapter will translate an analysis of the current state of affairs into a set of waste management goals and program objectives for the community.

Chapter Six - Planning a Strategy

Chapter Six will lead the user finally to a series of discrete alternative approaches and technical options that may be appropriate for the community's solid waste program. This section will assist the decision-maker to develop an agenda for program managers and analysts to further evaluate technical options and program approaches.

3.3 Step By Step Approach to Volume II

The technical/implementation portion of Volume II will be divided into ten chapters as follows:

Chapter One - Introduction
Chapter Two - Description of Disposal Options

Chapter Three - Selection of Approach
Chapter Four - Implementation of an Approach
Chapter Five - Operating Revenues and Financing
Chapter Six - Collection
Chapter Seven - Transfer Stations and Hauling
Chapter Eight - Processing
Chapter Nine - Commercial and Industrial Solid Wastes
Chapter Ten - Special Wastes

This arrangement of the chapters is based on the premise that the array of disposal options included in an integrated approach affects the collection, processing, and storage systems used by the community. As a result, this organization is meant to lead the local decision maker through the selection of disposal options first because this may affect later decisions associated with collection, storage, and processing.

An alternative arrangement would be to organize the chapters according to the flow of the waste (i.e., collection, storage and processing, and disposal), followed by the chapters describing the selection and implementation of an integrated approach. This arrangement is also reasonable, as it can be argued that the collection process may itself help to determine the array of disposal options available to the community (for instance, a rural community cannot undertake curbside source separation and recycling if trash is not collected at the curb). For the purposes of this exposition, the first arrangement was chosen because it is our view that the disposal options may more strongly influence decisions about collection, storage, and processing, and thus the discussion of the disposal options should precede these other areas.

Chapter One - Introduction

This chapter will accomplish the following tasks:

- Provide a brief overview of the solid waste crisis;
- Place this technical volume in context by briefly reviewing previous EPA activities and guides on integrated waste management; and
- Outline the purpose of this volume.

Chapter Two - Description of Disposal Options

The purpose of Chapter Two is to introduce each of the disposal options to the decision maker. The disposal options that will be covered are incineration, recycling, composting, landfilling, and source reduction. For each of these disposal options the chapter will provide a description of the various technologies (i.e. mass burn and refuse derived fuel incinerators, static pile composting, etc.) and will review important economic, environmental, and political/institutional issues associated with each of the technologies. This chapter is meant to provide decision makers with enough information so that they can make informed choices about the types of disposal options to include in an integrated approach.

Chapter Three - Selecting an Approach to Solid Waste Management

Chapter Three will assist local decision makers in selecting an approach to solid waste management. It will describe both the types of information that must be collected and criteria important to the selection of an integrated approach. It will then elucidate decision-making processes that communities can use in choosing an integrated approach.

Chapter Four - Implementation of an Approach

The purpose of Chapter Four is to provide a detailed description of how an integrated approach to solid waste management should be implemented. In essence, this chapter will provide technical managers with a step-by-step approach for implementing each of the waste management options chosen by the decision-makers.

Chapter Five - Operating Revenues and Long-Term Financing

Chapter Five will examine the various means of generating operating revenues and obtaining short and long term capital financing for solid waste projects. Although many of the issues discussed in this chapter will have been touched on in previous sections (such as capital financing for incinerators and landfills), there are a number of over-arching issues related to setting fees and creating economic incentives that are critical to the success of an integrated approach to solid waste management. The purpose of this chapter will be to cover these areas and present a detailed description of how communities can generate operating revenues and obtain capital financing.

Chapter Six - Collection

Chapter Six will discuss the collection of municipal solid waste.

Chapter Seven - Hauling and Transfer Stations

Chapter 7 will address the various methods of hauling waste to disposal sites and examine the use of transfer stations. Hauling methods that will be examined include trucking, barging, and potentially railing.

Chapter Eight - Processing

Chapter 8 will examine various methods of processing waste prior to disposal. The purpose of this chapter will be to introduce the various processing technologies to the decision-maker and indicate how they may be used in conjunction with the disposal options discussed previously.

Chapter Nine - Industrial and Commercial Solid Waste

Chapter Nine will describe special features about industrial and commercial solid waste that affects the methods used to collect, process, and dispose of this waste.

Chapter Ten - Special Wastes

Chapter Ten will examine the collection, transport, processing, and disposal of special wastes that may be present in the solid waste stream.

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THE TEN MYTHS OF MUNICIPAL WASTE COMBUSTION
AND
THE FUNDAMENTALS AND POLITICS OF GARBAGE

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ABSTRACT

Municipal waste combustion (MWC) is a technology that has been evolving for several decades throughout the industrialized world and is regarded by solid waste professionals and practitioners as an environmentally safe, cost effective strategy in solid waste management and disposal. However, it is a technology that is currently struggling to overcome an adversity which cannot be solved by engineers. Bad public perception. The general public has been led to believe that facilities utilizing MWC commonly known as resource recovery facilities (RRF) or waste-to-energy (WTE) facilities present an unacceptable risk to human health and should not be part of the 'solution' to solid waste management. These negative public perceptions are not based on fact or sound logic; they are rather the result of a campaign by volunteer organizations opposed to municipal waste combustion with social or political agendas.

This paper is an attempt to counter some of the sleight-of-hand logic, misleading information, and incorrect data so often spoken at public meetings and reprinted by the media regarding municipal waste combustion. It is also an attempt to introduce the concerned citizen, public official, and citizens task force member, to the definitions, logic, and information that permeates the discussions and clamor of the 'burning issue.' A background description of the political atmosphere involving the voting public, government officials, volunteer environmental activists, and the media is used as a stage set for presenting the logic and facts associated with 10 of the most common myths of MWC.

INTRODUCTION: THE POLITICS OF GARBAGE

INTEGRATED APPROACH - THE BLEND

The politics of garbage has given the nation the Integrated Approach. This is the 4-part hierarchy adopted by the EPA(1) and most state regulatory groups that says MSW management and disposal will best be achieved by a blend of solutions, some high-tech and some low-tech, with the priority:
1) waste reduction; 2) recycling; 3) waste-to-energy; and 4) land disposal. The integrated approach idea is a good one. However, the hierarchy is a political compromise that looks decisive but, alas, misses the mark by emphasizing solid waste management strategies that are not available to those who are responsible for disposal - local governments. The 4-part hierarchy would be fine if the federal government or even states were responsible for disposal because legislation could then be introduced to require waste reduction and create markets in the commercial sector for recyclables.

Everyone who has adopted the hierarchy seems to agree that the most important strategy is to reduce waste generation. But what is this excess waste we generate in America that can be eliminated? It's newspaper supplements, catalogues, junk mail, and packaging. But who's got the muscle to take on the media, advertising industry, and packaging industry? Not the city; and not the county. Therefore, everyone at the local level accepts reduction as number 1 on the hierarchy but ignores it. The politics of garbage.

For example, if we apply the hierarchy to a typical community of 350,000, listing the waste generators and prioritizing from worst to best, we would see that some businesses generate more solid waste in producing their product or service than others. Residents fit near the bottom which is not a political priority but a realistic one for a free economy. They buy what they want and throw away what they don't want. Because waste reduction is number 1, we decide to adopt some program that penalizes businesses for producing waste. The more they contribute to the solid waste problem, the more they pay for solid waste disposal. Perhaps more significantly, the more their product contributes to the solid waste problem, the more they contribute to paying for solid waste disposal. This concept gets into the useful life of packaging products sensationalized by the styrofoam hamburger container which has a useful life of a few minutes. A waste reduction program of this type would identify one business in our community that generates a product almost all of which is discarded as refuse within 24 hours. A prime target for waste reduction, yes? No. It is the local newspaper and even though its product contributes significantly to the waste stream both in quantity and composition (Myth 7) it is the only member of the community that "buys ink by the barrel." The politics of garbage.

HIERARCHY LOGIC

The 4-part hierarchy of solid waste management has been endorsed by most state regulatory agencies, environmental activist groups, the media, and general public. It has a lot of appeal and therefore is politically popular with governmental officials at all three levels (federal, state, and local) for two reasons: 1) It makes the mitigating statement that a particular technological approach is not necessarily correct but that a blend of technologies, if implemented in a prioritized fashion, will ensure the correct approach ("compromise is the art of politics"); and 2) It places the two most effective, yet unpopular strategies, waste-to-energy and landfilling, at the bottom while giving top priority to the least effective but most popular strategies. It is not surprising that this hierarchy was initiated at a level of government not responsible for solid waste disposal, in this case U.S. EPA. At the local government level where all of the responsibility for solid waste disposal lies, the hierarchy might be: landfilling, waste-to-energy, recycling, and waste reduction. This is the hierarchy that would give the cities and counties the greatest leverage to deal with the problem.

The logic is that only two of the four strategies can be effected by local government and only two are disposal strategies. They are the same two; waste-to-energy and landfill. The other two are solid waste management strategies that are intended to rectify the characteristics that make us the greatest waste generators in the industrialized world. And, assuming these strategies are intended to effect reductions in our solid waste stream rather than effect social change, they are worthy of our most sincere efforts. Unfortunately, they simply don't deliver much disposal solution for the dollar. But worse, the attention and priority they have received has led the general public to hope that if number 1 and 2 are implemented with full commitment, then number three and four might not be necessary.

Only Two Disposal Strategies: What is the difference between 'waste disposal' and 'waste treatment?' As analysis becomes more rigorous, the distinction between disposal and treatment fades, due to the fact that matter cannot be created or destroyed. However, material compounds or manufactured products are formed from fundamental elements and matter can be broken down or decomposed through decay or combustion. And whereas neither of these decomposition mechanisms actually destroys or eliminates material, the mechanics of decomposition achieves a significant reduction in the mass of the original material. In solid waste management, this reduction in mass or volume is the measurement of disposal and it can only be achieved through landfilling (including composting) and combustion. The other strategies, waste reduction and recycling, reduce the amount of material that must be disposed of by not creating it, or reusing it.

Landfills have always had a bad reputation with the public for obvious reasons, but they have always been accepted also for obvious reasons. Waste-to-energy technologies patterned on the European experience, rapidly developed in the U.S. in the 70's as a means to alleviate the diminishing landfill capacity problem. However, beginning in 1985, volunteer environmental organizations attacked the waste-to-energy approach as a MSW disposal strategy that was outdated and posed an unacceptable health risk to the public. Today, municipal waste combustion, and WTE or resource recovery facilities have a distorted reputation with the general public. As a matter of fact, it is so bad that this proven and effective strategy for MSW disposal, which is the main spring of solid waste management in Europe and Japan, has been categorically rejected by many U.S. urban communities even after millions of public dollars were spent to develop it. For these communities, the real alternative will not be the hopes and dreams of waste reduction, recycling, or social change, but rather large remote landfills with rail access. Perhaps the failing family farms in the Nation's heartland will be converted to large landfills with rail lines to provide solid waste disposal for the urban coastal communities.

PUBLIC PERCEPTIONS, MISCONCEPTIONS, AND EDUCATION

Assume you live in a typical U.S. community with a population of 350,000 in 1989. Your local government is legally responsible for the disposal of about 240,000 tons(2) of municipal solid waste per year, which translates to about 650 tpd (tons per day). You are a responsible citizen in this community. You pay your taxes, participate in the PTA or civic organizations, and perhaps attend a local church. You care about the quality of life and well-being of your citizenry. You consider yourself an environmentalist, meaning that you support pro-environment positions regarding the quality of air, water, land, and natural habitats. You read the local paper and watch the local and national evening news broadcasts.

Let us also assume that your local government has been studying solid waste disposal alternatives which include undesirable facilities such as new or expanded landfills and incinerators. You have formed some opinions, perhaps strong ones, about how local government should deal with this problem and you are concerned that your local officials are on the wrong track...perhaps overly influenced by the high-tech approach promoted by the expensive consultants they have hired. You are convinced that they have overlooked the low tech common sense aspects to solid waste (SW) management that every responsible citizen can plainly see. It is obvious to you that Americans are the greatest waste generators in the world; maybe the most wasteful society in general terms. You realize the need for landfills will continue in the foreseeable future, but they need to include greater safeguards against leachate migration and subsequent groundwater contamination.

However, recycling is an idea whose time has come, and you feel the government should not invest a lot of money in long-range high tech solutions until recycling is given a chance. It just makes common sense that reusing something is better than throwing it away and if we fully commit to recycling, perhaps the existing landfill will last long enough for the country to figure out a better way to dispose of solid waste. You're almost convinced that if your local government makes the maximum commitment to recycling, you won't need that 'incinerator' proposed by the consultants, but more importantly, you're sure that if the 'incinerator' is built recycling cannot be fully realized because recycling and incinerators are incompatible. Therefore, the underlying question you see facing your community and your local officials regarding solid waste disposal in the future is: "Do we burn or do we recycle?" You've also been to public hearings and listened to all kinds of data and rationale from officials and experts that recycling won't solve the problem because of markets and institutional barriers and things that you don't fully understand; but, nevertheless seem like things that people and government can solve as opposed to toxic chemicals poisoning the air, water, and land. You may not understand all the technical details, but the fundamental logic is clearly 'burn or recycle.' High tech vs. low tech. Experts vs. the people's common sense.

Rejection of Combustion: The above attempts to typify the position of the U.S. citizen who is active in municipal solid waste issues. Municipal waste combustion has a terrible reputation with the public. The underlying reason for this is lack of public education regarding solid waste issues, but that's like Will Rogers' solution to the U-boat problem in WWII. "Boil all of the water out of the Atlantic ocean so you can see 'em."

How does the greater technical community involved in the problems and solutions regarding MSW management and disposal educate the public about the facts regarding environmental impacts, costs, legalities, responsibilities, and free market forces that pertain to ALL solid waste management and disposal alternatives? The citizen above got most of his information from the print and broadcast media who are quick to state that they are not in the business of public education. The media does not teach about the issues; it reports the news. This position may be correct but it is frustrating to local governments trying to compete with the editorial page.

Those opposing a local government initiative have the advantage of not being required to substantiate the validity of their comments while government officials and consultants are professionally required to be able to substantiate all decisions, positions, and recommendations. When the issue or initiative is technically sophisticated, expensive, or undesirable, (MWC facilities are all three) the editorial page dilemma is particularly bad because of the requirement for brevity. Someone opposing a municipal waste combustion (MWC) facility on the editorial page, is free to use emotionally punchy language that preys on the reader's ignorance in the matter and crystallizes negative images. It provides a wonderful podium for clever, usually incorrect, statements that arouse the anxiety ghosts in our environmental closets. For example, referring to a modern municipal waste combustion facility as an incinerator, garbage burner, or burner, and stack emissions as 'landfill in the sky' or changing the name of dioxin to dioxin-the-most-toxic-substance-known-to-mankind.

The responsible response to these images is typically a position or discussion, based on facts, figures, and related references that attempt to educate the reader about the issues, logic, complexities, trade-offs, and decisions by his government. These responses are too comprehensive for the editorial page, and the reassuring arguments by government officials which are brief enough for the editorial page lack the emotional language and imagery. When the mayor, city manager, or councilman says things like "we have carefully studied all the options...our decisions have been approved by appropriate federal and state regulatory people...we are replicating successful experiences elsewhere utilizing the proven technologies available to us...our comprehensive environmental studies and citizen task force report indicates that no unreasonable risk to public health will be incurred...." The reader thinks: "Sure, sure, sure...but what about the landfill in the sky?" "What about a trillionth of a gram of dioxin killing a pregnant monkey?" And when the local government, who is responsible for the collection and disposal of solid waste conducts a public meeting to educate the citizenry, the majority can't or don't attend and the local press covers the event rather than the information presented. Foreseeing this scenario, the local government wisely video-records the presentation and broadcasts it over the local cable public access channel. But if they broadcast it outside of prime time the people they are trying to reach will be asleep, at work, or out of the house. And if they broadcast it during prime time they compete with everyone else's prime time broadcasts.

The public education problem is that most of the people don't want to be educated about boring stuff like what to do with garbage.

Siting and NIMBY: Any discussion of the politics of garbage must include the NIMBY (not-in-my-backyard) phenomenon. NIMBY is a political facet that affects the siting of all undesirable public facilities including sewage treatment plants, power plants, fire stations, new roads, airports, jails, mental patient quarters or clinics, homeless shelters, and solid waste treatment, handling or disposal facilities such as landfills, resource recovery plants, transfer stations, and recycling centers. However, unlike the others where there is at least some segment of the population that sees a benefit or need for the proposed facility, solid waste disposal facilities have no perceived benefit by the public. This is due in large part to the long standing practice by municipal governments of providing biweekly residential curbside pickup. The trash has been magically disappearing for so long that it is difficult to suddenly get the public to address the technical, financial, and environmental aspects and tradeoffs involved in solid waste management and disposal. The public wants their garbage picked up as usual and for the most part accepts that the local government has to build disposal facilities that must be paid for with local revenues, so long as they are 'not in my backyard.'

The siting issue is politically powerful because it impacts neighborhoods where political opposition is fairly easy to organize so long as the target is a common enemy such as city hall. But it is particularly powerful in the case of solid waste disposal facilities because it transforms the average citizen into an 'environmentalist' who is suddenly more concerned about air and water quality than anything else. This combination of neighborhood, city-hall-target, and environment makes the siting of solid waste disposal facilities perhaps the most politically difficult task a local government can undertake. The same guy who would slam the door in your face if you asked him to support opposition to wood stoves, barbecue grills, and vehicle exhausts - all of which pose a greater threat to his air quality than a modern MWC facility - is now an environmentalist with grave concern about the air quality.

MYTHS VS. FACTS: THE BATTLE OF PUBLIC PERCEPTION

ENERGY CRISIS IN THE MID 1970'S

This may not appear to relate to municipal waste combustion; however, as a technology-based crisis this national phenomenon had a lot in common with other technology-based crises such as nuclear power and municipal waste combustion and the lessons learned directly apply. The public was convinced that we had finally reached the limit of petroleum resources, and some believed we were running out of energy, period. The political movement followed public perception and the U.S. was suddenly possessed with an energy conservation theme, which provided fertile ground for inventors, some earnest and some promoting snake oil, but both with their feet planted firmly in midair. Remember the network evening news broadcasts that would show us long lines at the gas stations followed by a story about a guy who had invented a way to get a hundred miles per gallon as demonstrated by a device on his lawn mower?

This crisis mentality also resulted in many otherwise reasonable middle class Americans thinking it made more sense to heat their homes with a wood stove or kerosene heater than to depend on the utilities. Many not only bought wood stoves, but chain saws and pick-up trucks so they didn't have to depend on wood suppliers either. It was a matter of low tech, common sense and self-reliance.

Logic, Facts, and Lessons: After a couple of centuries of accepting the fundamental principle that energy, like matter, cannot be created or destroyed, how could we suddenly have an 'energy crisis?' What we had was a fuel market crisis, which may or may not have been contrived by some pretty big players and brokers such as OPEC. Maybe it was a fuel location 'crisis.' Through depletion, petroleum resources had shifted to an area of the world that was less and less controlled by the U.S.. The 'crisis' should have awakened America to the realities of geography, politics, and oil economics, namely, that we could not continue to use finite resources belonging to other countries without discretion. However, the 'crisis' fostered social awareness of the fact that we were the world's greatest consumers of energy. A national energy policy should have resulted, but instead, our shame lead to 'energy conservation' as the theme for about 10 years. But, we now know that a decade of relatively cheap and plentiful oil has lulled us back into muscle cars and new ways to finance highway construction. Meanwhile, in Western Europe, a gallon of gasoline costs three times more than it does in the U.S.

(just as it did before the 'energy (fuel) crisis') and financing of new transportation systems is probably more likely to be for high speed rail than expressways.

There are many similarities between the 'energy crisis' experience and our current solid waste disposal crisis. More specifically, the crisis psychology is being used as a lever to effect social change in the way America regards the generation and disposal of municipal solid waste. Whether or not these changes are meritorious is not a technical argument and should be left to the makers of public policy. However, one of the casualties of this current crisis is the incorrect public perception that modern municipal waste combustion facilities are an unacceptable strategy or tool in solid waste management. Municipal waste combustion has been and continues to be a primary strategy for environmentally sound solid waste management in Europe and Japan. It is a technology that has continued to develop with each new facility employing the latest design features for environmental controls, energy recovery, and materials recycling. It is a technology that represents a demonstrated tool for local governments who are responsible for the management and disposal of municipal solid waste.

What follows is a series of discussions intended to educate the concerned citizen, local elected official, community leader, taskforce or advisory group member, and the media, regarding the facts, perceptions, and misconceptions surrounding MWC as it applies to waste-to-energy and resource recovery facilities. These discussions are framed as ten myths of municipal waste combustion and are deliberately presented in a casual, common sense tone devoid of as much technical jargon as possible. All data and technical information is either referenced to indicate validity or is taken from the project files of the author.

MYTH NO. 1: AMERICA CURRENTLY FACES A 'LANDFILL DILEMMA' AND SOLID WASTE DISPOSAL CRISIS.

There is a crisis, but it is not a solid waste disposal or landfill crisis in the sense that we do not have the means and resources to dispose of our solid waste. The crisis relates more to public education, NIMBY (Not-in-My-Backyard), and cost of disposal. Cities and metropolitan areas in America are rapidly running out of landfill capacity and are facing tipping-fee-shock at remote rural landfills. As urban landfills become 'landfulls,' the long running conflict between urban and rural communities over WHO is paying a fair share for WHAT government service and quality of living, is now manifested in solid waste disposal. The cry for regional solutions is usually initiated by the cities.

Indeed, the issue is not landfill capacity. Compared to other industrialized nations with much greater population densities like Germany and Japan, the U.S. has plenty of space to construct landfills, but nobody wants them located in their backyard. The underlying issue regarding the so called 'disposal capacity crisis' is urban vs. rural. The urban populations want more parks and recreational facilities so they can feel like they are living in the country. They want the airport, jails, factories, and waste disposal facilities in the boondocks - out of sight, out of mind. Rural regional landfills with rail access are now being implemented or planned in several areas of the country. It's just a matter of time before the rural populations demand that urban populations deal with urban solid waste with more aggressive strategies than source separation and recycling centers.

Resource recovery facilities have been operating at a number of inter-city sites for at least a decade producing energy products such as steam, electricity, and chilled water to nearby customers. Perhaps a fair trade off for regional solid waste management would be to locate the high tech facilities (waste processing and combustion) in the cities where most of the waste is generated, and locate the landfills in the country. The landfills, of course, will be located in rural areas regardless, but with regional cooperation, they would be smaller and less threatening to the environment.

MYTH NO. 2: THE EUROPEANS AND JAPANESE ARE AHEAD OF THE U.S. IN DEALING WITH SOLID WASTE BECAUSE THEY GENERATE LESS AND RECYCLE MORE.

Table 1 indicates that the West Germans and Japanese who are technologically advanced societies with far greater population densities than the U.S. recycle a greater percentage of their waste stream.

They also generate less solid waste per capita-day and reuse more of their solid waste stream. (The distinction between 'reuse' and 'recycle' often obfuscates reported 'recycling' percentages). The myth here is the assumption that these two reasons alone account for their advanced degree of management. As indicated, the most significant difference between the U.S. management strategies and the others is the amount of solid waste that is landfilled. They landfill less because they process more at combustion or WTE facilities.

TABLE 1. SOLID WASTE MANAGEMENT IN THE UNITED STATES, JAPAN, AND WEST GERMANY (%)(3)

	U.S.	JAPAN	W. GERMANY
Recycled/Reused	10	50	15
Waste-to-Energy	5	23	30
Landfilled/Other	85	27	55
TOTAL	100	100	100

As for the perception that municipal waste combustion and waste-to-energy facilities are becoming obsolete in Europe and Japan, again the figures for West Germany are probably a good indicator of the current and future trends in Europe. Table 2 indicates how these two countries dispose of the MSW left over after recycle, and the ratio of combustion to landfill is expected to increase in both Europe and Japan. In the fall of 1988, the West German Environmental Protection Agency announced[4] the implementation of a new program to construct 20 to 25 waste-to-energy facilities by the year 2000, that would combust about 4.4 millions tons per year of solid waste.

TABLE 2. DISPOSAL OF UNRECYCLED SOLID WASTE IN THE UNITED STATES, JAPAN, AND WEST GERMANY %[3]

	U.S.	JAPAN	W. GERMANY
Waste-to-Energy	6	46	35
Landfilled/Other	94	54	65
TOTAL	100	100	100

MYTH NO. 3: GARBAGE INCINERATORS, OR TRASH BURNERS, ARE YESTERDAY'S TECHNOLOGY, AND BURNING MSW IS BECOMING OBSOLETE.

Incinerators ARE yesterday's technology. For several decades the term 'municipal incinerator' referred to those brick buildings with brick chimneys that 'burned garbage to ashes', which is the American definition of 'incinerator.' Traditionally, the term 'combustion' has referred to the chemical reaction involving heat, fuel, and air that breaks down the fuel matter at a molecular level resulting in the production of 3 things: 1) exhaust gases which are referred to as the products of combustion; 2) ashes which is the inert residue left; and 3) heat.

If you were to observe a simple incinerator, you would see unprepared MSW (garbage) being fed into a refractory brick furnace. You would also see hot gases being emitted from the chimney and hot ashes being removed from the grate, or furnace floor. In short, you would see a combustion process which is typically not very different from the fireplace or wood stove in your house. And, because the fuel and air feed rates of the incinerator are regulated manually with little means to mix them together during combustion, this process would be referred to by combustion engineers as 'uncontrolled combustion.'

And because there is no heat recovery or cooling of the hot (1800 F-2000 F) furnace exhaust gases, there is no gas cleaning or air pollution control equipment. Due to the design features of air pollution control equipment, the exhaust gases must be in the 400 F to 800 F range in order to be processed or cleaned, which is why modern MWC is synonymous with WTE. That is, you must recover the heat (energy) in order to clean the gases.

Modern industrial and utility grade furnace/boilers include an array of automatic controls to regulate the combustion process for thermal efficiency and air quality of emissions. Primary controls include: rate of change of fuel feed and combustion air; the ratio of fuel and air; turbulence (mixing); and temperature of the exhaust gases as they exit the furnace. Secondary or 'trim' controls fine tune the primary controls to minimize the formation or emissions of organic pollutants by continuously monitoring indicators of complete combustion such as oxygen, carbon dioxide, or carbon monoxide. It would be incorrect and misleading to refer to this 'controlled combustion' as 'incineration,' and because it involves the mechanics and chemistry of combustion, the technical community has adopted the term, 'combustion' to also describe the process (e.g. a coal fired combustion unit or facility; oil/gas fired combustion system). Therefore, if one were to burn MSW in a furnace/boiler that included all the combustion control features of a modern industrial boiler, it would naturally be referred to as 'municipal waste combustion' (MWC), which is very different from the obsolete uncontrolled garbage 'incinerator.'

There are some technical distinctions between modern MWC units and modern industrial boilers. In addition to the primary and secondary automatic control systems, modern MWC furnace-boilers have special configurations (shapes) to provide a predetermined time-temperature profile in the furnace. That is, the exhaust gases are exposed to a minimum temperature for a minimum amount of time with the correct amount of air mixture to ensure complete combustion of the various compounds in the gas stream. Additionally, the grate system is independently controlled or modulated to move the refuse through the furnace to provide the complete combustion or transformation to ashes. Exposure time in the furnace, continual mixing of the burning material with air, and continuous trimming of the combustion parameters are the key control features that are intended to minimize organic pollutant emissions from the furnace and distinguish these units from fossil fired combustors. Of course, material which does not combust such as glass and metals passes through, however, even noncombustible material is at least sterilized by exposure to the furnace environment. That is, combustible compounds attached to the incombustible material will be burned or destroyed. For instance, the paint or wrapper on a soup can, the bacteria in a food jar, and the toxin residue in a can of oven cleaner will be combusted even though the can or jar is not.

The other major difference between modern MWC units and industrial boilers is the air pollution control system that cleans the exhaust gases from the furnace. MWC facilities typically incorporate gas scrubbers that remove 70% of the SO2 and 90% of the HCL from the exhaust gases as well as high efficiency particulate, or dust, collectors in the form of fabric filters or electrostatic precipitators that remove 99+% of the suspended particles. On a heat output basis, modern waste-to-energy facilities emit less pollution than industrial and utility plants except for those which burn only natural gas.

Figure 1 is a cross section of a typical 1960's vintage municipal incinerator, many of which were built and operated for years in areas of high population densities such as the Northeast and Great Lakes industrial regions. Figure 2 is a cross section of a waste-to-energy facility employing modern mass burn MWC, and Table 3 is a comparison of the key design features of the two systems. If we call Figure 1 an incinerator, we have to call Figure 2 something else. Resource recovery facility? Waste-to-energy facility? Municipal waste combustion facility? Perhaps it depends on the emphasis of the application, but regardless of the preferred designation, comparing the combustion technology and the level of sophistication between them is like comparing a modern automobile to a Model A Ford. Of course, some will say the Model A Ford was better.

Today's MWC facilities are NOT 'incinerators,' and anyone who insists on referring to them by that name is either not informed or inherently biased, either of which diminishes their credibility in the solid waste management forum.

MYTH NO. 4: RESOURCE RECOVERY FACILITIES CONSTITUTE A NEW OR ADDITIONAL ENVIRONMENTAL PROBLEM AS OPPOSED TO A SOLUTION TO AN ENVIRONMENTAL PROBLEM.

Experienced solid waste engineers view the emergence of MWC facilities as a means to improve and extend the life of sanitary landfills, but, the public doesn't perceive it this way. In effect, when the public directly rejects resource recovery or municipal waste combustion as 'environmentally unacceptable' it is indirectly endorsing the other disposal alternative, landfill. And whereas nobody wants a new landfill in their backyard, the public by and large has indicated throughout the land that landfills are preferable to resource recovery especially if they have lots of liners, leachate treatment, and screening; and are located somewhere else like the boondocks. This preference has been reconfirmed by the public misconception that biodegradable products are environmentally desirable.

Perhaps the general public is more comfortable with landfills than they are with resource recovery facilities because: they have been with us forever; they're low tech which is not as threatening as high tech; and they are not as visible at ground level as a 100 ft. high building attached to a 250 ft. high stack. Landfills, however, are a threat to the environment and human health, more so today than in the past due to the chemically sophisticated composition of today's waste stream. But how do you compare the environmental impacts of landfill vs. combustion? The answer is total toxic emissions to the air, water, and ground. For the typical community with a population of 350,000, let's compare the quantities and toxicity of the disposal of 650 tons of solid waste generated daily.

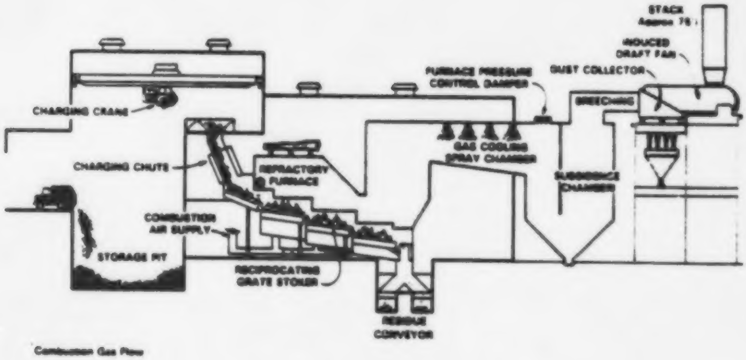
STACK
Approx 75'
INDUCED DRAFT FAN
DUST COLLECTOR
FURNACE PRESSURE CONTROL DAMPER
BREECHING
CHARGING CRANE
GAS COOLING SPRAY CHAMBER
CHARGING CHUTE
REFRACTORY FURNACE
SUBSIDENCE CHAMBER
COMBUSTION AIR SUPPLY
STORAGE PIT
RECIPROCATING GRATE STOKER
RESIDUE CONVEYOR
Combustion Gas Flow

Figure 1: TYPICAL 650 TPD INCINERATOR (Circa 1960)

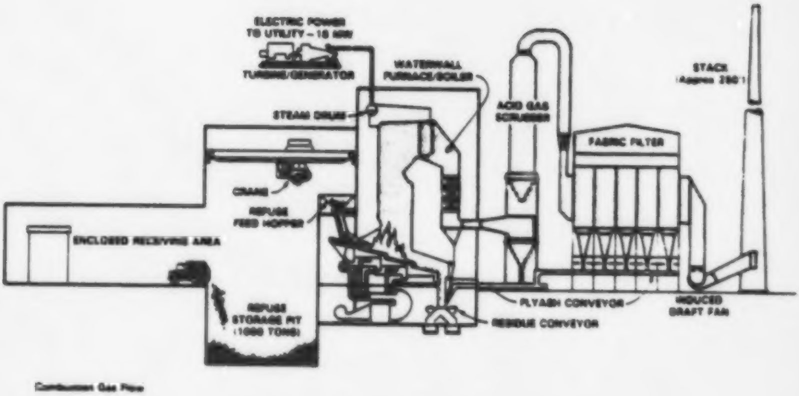
ELECTRIC POWER TO UTILITY – 18 MW
TURBINE/GENERATOR
WATERWALL FURNACE/BOILER
STACK
(Approx 250')
STEAM DRUM
ACID GAS SCRUBBER
FABRIC FILTER
CRANE
REFUSE FEED HOPPER
ENCLOSED RECEIVING AREA
FLYASH CONVEYOR
INDUCED DRAFT FAN
REFUSE STORAGE PIT (1000 TONS)
RESIDUE CONVEYOR
Combustion Gas Flow

Figure 2: MODERN 650 TPD MUNICIPAL WASTE COMBUSTION FACILITY WITH ENERGY RECOVERY (1988)

TABLE 3. COMPARISON OF KEY DESIGN FEATURES OF TYPICAL 1960 INCINERATOR AND 1988 WASTE-TO-ENERGY FACILITY

DESIGN FEATURE	INCINERATOR	MODERN WTE FACILITY
Tipping floor	Exposed	Enclosed
Refuse storage pit	Yes	Yes
Storage area odor control	No	Yes
Charging crane control	Local;Visual	Remote;Monitor
Refuse feed	Cascade	Auto. Ram Metering
Reciprocating grates	Manual;Independent	Automatic;Integrated
Primary combustion air	Manual;Independent	Automatic;Integrated
Overfire air	Manual;Independent	Automatic;Integrated
Furnace design	Small;Low Cost	Large;Time Temp Profile
Heat recovery	None	Electricity;Steam
Acid gas removal	None	90% of HCL;70% of SO_2
Organic emission control	None	Automatic;Continuous
Combustion control system	None	Auto at 3 levels
Particulate removal	50 - 75%	99.+%
Stack height	50 - 100 ft.	200 - 300 ft.
Air pollution control system	< 5%	20 - 30%
Environmental testing	None	Periodic & comprehensive
Emissions monitoring	None	Continuous

Landfill: If we bury 650 tons per day of typical MSW, the disposal process will be decomposition which is slow, and the emissions will be in the form of leachate discharge to the ground and methane gas discharge to the air at ground level. The control strategy for leachate is collection and treatment and, for the methane gas, it is underground collection and vent or combustion. Both of these emissions are delayed due to the nature of decomposition, but over a 10 to 20 year period the landfill will generate about 8,400 cubic feet of methane gas for each ton of MSW buried, and about 375 gallons of leachate per acre per day based on a percolation rate, or rainfall, of 5 inches per year. Therefore a single daily load of 650 tons will eventually produce about 5.5 million cubic feet of methane gas.

In addition to discharges and emissions, landfills typically have problems with vectors and/or seagulls which may not constitute an environmental threat but are nevertheless a nuisance to neighboring communities.

Combustion: Remember the 'landfill in the sky' hyperbole? Table 4 quantifies the material that goes in and comes out of a 650 tpd municipal waste combustion facility.

TABLE 4. MATERIAL BALANCE FOR TYPICAL 650 TPD MUNICIPAL WASTE COMBUSTION FACILITY

Material INTO Combustion System:

	LBS/HR.	TONS/DAY	TONS/YEAR
MSW:	54,166	650	204,748
AIR:	377,090	4,526	1,425,402
TOTAL:	431,260	5,176	1,630,150

Material OUT OF Combustion System:

	LBS/HR.	TONS/DAY	TONS/YEAR
RESIDUE/ASH:	14,880	179	56,230
STACK GAS EMISSIONS:	416,380	4,997	1,573,918
TOTAL:	431,260	5,176	1,630,150

Simply stated, MSW and air are introduced into the furnace where combustion takes place; and, exhaust gases, ashes, and heat are emitted from the furnace/boiler system. The material coming out must add up to the material going in. Heat is not material and is a by-product in this case, although 'bonus' would be a better term because the heat has a market value of $2 to $5 per million BTU's which translates to about $20 to $50 in revenue for each ton of MSW processed by a WTE facility.

Table 5 presents a breakdown of the composition of the exhaust gases. The combustion of 650 tons of MSW results in the emission of about 5000 tons of gases from the stack, but 99% of these emissions are non-pollutants. The remaining 1% that are pollutants are sub-divided into the 7 criteria pollutants and 14 non-criteria pollutants established by EPA regulations and technical guidance. The pollutant that gets the lion's share of the attention is dioxin, which even on a tons per year basis is barely measurable at .00000000013% of the total emissions.

What about the CO_2 emissions and the 'greenhouse effect?' In the case of WTE facilities, there is little or no net contribution of CO_2 to the atmosphere because the energy recovered from the WTE offsets fossil fuel combustion somewhere else.

TABLE 5. COMPOSITION OF STACK EMISSIONS FROM 650 TPD MUNICIPAL WASTE COMBUSTION FACILITY

			LBS/HR.	TONS/YEAR	% BY WEIGHT OF TOTAL STACK EMISSIONS
CRITERIA POLLUTANTS:					
Particulates	PM	:	6.50	24.75	.0016
Sulfur Dioxide	SO_2	:	38.48	145.45	.0092
Volatile Organics	VOC	:	1.48	5.57	.0004
Nitrogen Oxide	NO_x	:	155.72	588.58	.0374
Carbon Monoxide	CO	:	137.02	517.94	.0329
Lead	Pb	:	1.48	5.57	.0004
NON-CRITERIA POLLUTANTS:					
Sulfuric Acid Mist	H_2SO_4	:	6.160	23.26	.0015
Hydrogen Chloride	HCL	:	24.72	93.46	.0059
Hydrogen Bromide	HBr	:	1.672	6.32	.0004
Cadmium	Cd	:	.040	.054	.00001
Antimony	Sb	:	.122	.459	.00003
Arsenic	As	:	.006	.024	.000001
Mercury	Hg	:	.286	1.08	.00005
Beryllium	Be	:	$.174 \times 10^{-3}$	$.655 \times 10^{-3}$	.00000004
Fluoride	HF	:	.40	1.47	.0001
DIOXINS (USEPA EQ.)			---	2.1×10^{-6}	.0000000013
TOTAL AIR POLLUTANTS:			374.08	1,414	.0898%
NON-POLLUTANTS:					
Nitrogen	N_2	:	284,174	1,074,178	68.25
Oxygen	O_2	:	38,322	145,614	9.25
Carbon Dioxide	CO_2	:	56,244	212,602	13.51
Water	H_2O	:	37,066	140,110	8.90
TOTAL NON-POLLUTANTS		:	416,006	1,572,504	99.910%
TOTAL STACK EMISSIONS		:	416,380	1,573,918	100.00%

MYTH NO. 5: THE RESIDUE/ASH FROM MWC FACILITIES IS A "HAZARDOUS WASTE."

This is a trick question if there ever was one. The public perception is that a 'hazardous waste' is something that is hazardous to the environment and to human health. It's not that straight forward.

'Hazardous waste' is a legal term that is applied to a substance when its disposal must be regulated in accordance with Subtitle C of the federal regulations for the Resource Conservation and Recovery Act (RCRA). Another part of the regulations, Subtitle D, addresses the disposal requirements of municipal solid waste in sanitary landfills. RCRA and its state counterparts include a 'list' of compounds designated as 'hazardous waste.' If it's on the list, it's 'hazardous.' If it's not on the list, it doesn't mean it's not hazardous; and some hazardous wastes are specifically excluded from the list and from consideration under Subtitle C. Notably, in this issue, household wastes regardless of the composition or toxicity, are excluded.

Solid wastes which are not on the list and are not excluded can be designated a 'hazardous waste' by failing one of four EPA prescribed tests, one of which is known as the Extraction Procedure Toxicity, or 'EP Tox' test. This is a laboratory procedure that attempts to simulate the extraction or leaching of toxic compounds from the solid waste in question if it were buried in a landfill.

The residue/ash generated by a MWC facility is about 15% by volume of the raw MSW. Currently, there is a legal controversy at the federal and state level as to whether this material is subject to testing. Some EPA officials and some states (NY) have made the legal determination that residue/ash is covered by the language that excludes household waste. However, in those states which have not excluded it, MWC facilities face the challenge of having the residue/ash subjected to the EP Toxicity test to determine if toxic compounds such as heavy metals and dioxins will leach out. Many experts argue that the 'EP Tox' test is not fair because it does not replicate what actually happens at a real landfill, or monofill, containing this material.

When the EP Tox test is properly performed in accordance with EPA procedures on residue/ash from MWC facilities, all, or almost all, of the time it passes. When it fails, it is because the amount of lead or cadmium that is extracted by the leaching fluid exceeds a threshold value. Remember, lead and cadmium are metals which do not burn and are not products of combustion. They are present in the residue/ash only because they were present in the waste stream. It is true that the metals are more concentrated in the residue/ash because of the volume reduction provided by combustion, but the concentration argument is like saying that the solution to pollution IS dilution. In terms of threat to the ground water, the issue is how much toxic waste leaches out, and as indicated by Table 6, the actual values measured for both metals are higher for landfill leachate than they are for the ash leachate.

The EPA procedures for conducting this test are described in a document known as SW846, which is about 2 inches thick, and requires an experienced laboratory to perform. Most, if not all, of the reported EP Tox failures on MWC residue/ash were not conducted in accordance with SW846 and are therefore not valid. Unfortunately, the invalidity of information does not prevent it from being casually reported by the media. Remember, the EP Tox test cannot be performed on MSW or leachate from MSW landfills, because it is legally excluded from consideration. Therefore, regardless of how hazardous it may be to the environment, it is legally 'non-hazardous.'

Other tests have been performed to compare the toxicity of leachate from MSW landfills with the leachate from ashfills, and the indications to date are that the ash leachate is considerably less threatening to the groundwater than landfill leachate. Tables 6 and 7 summarize the data for comparative tests conducted by the environmental engineering firm of Malcolm Pirnie at the Westchester County, NY landfill. As indicated by the values in Table 6, the landfill leachate poses a greater threat to the groundwater than the ash leachate. Also notice in Table 7 that leachate from the ashfill exceeded USEPA Drinking Water Standards.

The reason dioxin concentrations in ash leachate are so small is because dioxin molecules bond very aggressively to particles, particularly inert particles like ash. This also helps to explain why dry scrubbers reduce dioxin emissions from the stack. When the lime slurry in the scrubber reacts with the acid gases, it flashes to a particulate in the process. This particulate is later removed by the dust collector, but in the meantime it provides a greater opportunity for the dioxins to attach. And, because it is very difficult to detach or wash the dioxins from the particles the dioxin stays in the landfill with the ash.

Additional tests have been conducted in this area, and EPA has a testing program under way to examine the toxicity of ash leachate, so, hopefully, the environmental aspects of the hazardous ash issue will soon be resolved.

It is also hoped and anticipated that the legal aspects of this controversial issue will be cleared up in the 101st Congress. In the meantime, it is ludicrous for anyone to suggest that a modern MWC facility somehow transforms 'non-hazardous' MSW into a 'hazardous waste' simply because the lead and cadmium in the MSW is temporarily more concentrated in the residue/ash causing some samples to slightly exceed regulatory thresholds, sometimes. Nevertheless, technology in a free society never sees "problems," just opportunities. Because it is profitable to find a way to treat residue/ash such that it doesn't fail the EP Tox test, or its successor, it will probably be available as an additional feature on future facilities.

TABLE 6. ASH LEACHATE VS. SANITARY LANDFILL LEACHATE (MILLIGRAMS PER LITER)(3)

LEACHATE PARAMETER	PILOT TEST WESTCHESTER COMBINED RESIDUE/ASH LEACHATE RANGE OF VALUES	TYPICAL SANITARY LANDFILL LEACHATE
pH	10.2 - 10.6	
TDS	3.90 - 15.900	1,000 - 10,000
TSS	2 - 30	100 - 400
Alkalinity, as $CaCO_3$	55 - 178	500 - 6,000
BOD	31 - 250	2,000 - 15,000
COD^5	102 - 487	5,000 - 40,000
TOC	25 - 152	1,000 - 10,000
Aluminum	0.4 - 1.5	NR
Arsenic	<0.005	<0.003 - 0.03
Barium	<0.2	ND - 1.1
Cadmium	0.01 - 0.05	<0.05 - 0.1
Chromium, total	<0.01 - 0.12	ND - 0.08
Chromium, hexavalent	NR	NR
Copper	0.25 - 0.64	ND - 9.0
Iron	0.04 - 0.10	10 - 1.000
Lead	<0.05 - 0.13	0.02 - 1.0
Magnesium	0.8 - 1.52	100 - 500
Manganese	<0.01 - 0.02	0.1 - 50
Mercury	<0.005	NR
Nickel	0.06 - 0.14	10 - 1.1
Selenium	<0.005- 0.009	0.02 - 1.0
Silver	<0.02	100 - 500
Zinc	0.006	0.1 - 50
Calcium	87 - 105	60 - 2,500
Chloride	163 - 5,100	100 - 2,400
Potassium	70 - 835	50 - 1,000
Sodium	155 - 3,360	100 - 4,000
Sulfate	1,760 - 4,330	25 - 500

TABLE 7. ASH LEACHATE VS. WATER QUALITY STANDARDS (MILLIGRAMS PER LITER)(5)

Water Quality Parameter	Pilot Westchester Combined Ash Residue Leachate	EP Toxicity Test Criteria	NYS "GA" Groundwater Standard	NYS Groundwater Discharge Standards	Port USEPA Drinking Water Standards	Washington WPCD Sewer Use Ordinance
pH	10.2 - 10.6	<2 or >12.5	6.5 - 8.5	6.5 - 8.5	6.5 - 8.5	>5.5, <9.5
Aluminum	0.4 - 1.5	5.0		2.0		1.2
Arsenic	<0.005	100.0	0.025	0.05	0.05	0.25
Barium	<0.02	1.0	1.0	2.0	1.0	2.0
Cadmium	0.01 - 0.05	5.0	0.01	0.02	0.010	0.1
Chromium, total	<0.01 - 0.02				0.05	0.25
Chromium Hexavalent		0.05	0.10			
Copper	0.25 - 0.64		1.0	1.0	1.0	0.4
Iron	0.04 - 0.10		0.3	0.6	0.3	1.5
Lead	<0.05 - 0.13	50	0.025	0.5	0.05	0.1
Manganese	<0.01 - 0.02		0.3	0.6	0.5	2.0
Mercury	<0.005	0.2	0.002	0.004	0.002	0.10
Nickel	0.06 - 0.14			2.0		2.0
Selenium	<0.005 - 0.009	1.0	0.02	0.04	0.01	0.10
Silver	<0.02 - 5.0	0.05	0.10	0.05	1.05	
Zinc	0.006 - 0.031		5	5.0	5.0	0.6
Chloride	163 - 5100		250	500	250	500
Sodium	155 - 3360					500
TDS	3090 - 15,900			1000	500	

MYTH NO. 6: IF WE FULLY COMMIT TO RECYCLING WE MAY NOT NEED MORE LANDFILLS OR MUNICIPAL WASTE COMBUSTION FACILITIES.

The second strategy on the EPA hierarchy is recycling and a large measure of the misconceptions about the effectiveness of recycling is vocabulary and definition, beginning with 'municipal solid waste.' Is MSW the total waste stream to be managed by the integrated approach, or is it the material that must be disposed of after reduction, recycling, and reuse?

Assume the local government of our 350,000 pop. community commits to recycling as the preferred strategy for solid waste disposal. How effective could that be? Or, how much disposal could be achieved? Assume that mandatory source separation is implemented and that every waste generator served by the government is required to sort his or her solid waste into the following components: aluminum, ferrous, glass in three separate colors, newsprint, magazines, corrugated paper, plastic beverage bottles and miscellaneous plastic. That's ten components that you place on the curb in ten separate receptacles to be picked up by a special truck that has the same ten separate compartments. The trucks haul the recyclables to a central recycling center owned and operated by the local government. Also, let's assume that yard waste such as leaves, grass, and trimmings are picked up separately once a week or bimonthly and that tires and batteries incur large deposits so that everyone returns them for redemption. What's left, the garbage, or MSW is picked up and hauled to the landfill in the typical manner. Now, assume that everyone participates in the program (100%) and nobody cheats or makes a mistake. What percentage of the 650 tons per day of solid waste would be recycled? The answer is zero! What has been achieved by the source separation and the city's separate collection is segregated solid waste. The percentage figure you estimated to the question applies to the amount of solid waste stream that would have been recovered or reclaimed for recycling. Not until someone in the commercial segment reuses the recovered material in the production of a new product has recycling been achieved.

The actual recycling of any constituent in the waste stream means the reuse or re-manufacture of that material in the making of a new product over and over. If it's recycled only once, then it's delayed garbage. For example, recycling something like plastic soda bottles into flower pots or paint bristles may eventually be feasible, but what happens to the old flower pots and paint bristles? How many times can newspaper stock be recycled into newspaper stock or something else? (So many of the waste paper recycling figures are deceiving because they include high amounts of exports to the third world.) With the exception of glass, metals and some plastics which are melted and reformed into new copies of the same product, the feasibility of continuous recycling of solid waste is suspect at the fundamental level. Incidentally, this should not be confused with reuse of items which are returned directly to the reuser such as bottles and containers.

So, the popular term 'recycling' really means 'recovery-reuse' a two headed beast which transcends the public/private segments of our socioeconomic system and is far more complicated than the simple idea suggested by the simple and positive term 'recycling.' More importantly, the public expects the local government to be responsible for implementing recycling in spite of the fact that the local government's authority ends with recovery and collection.

Municipal Service vs. Commerce: If there really is an underlying issue or question in the solid waste management arena it is probably this: Should the local government address solid waste disposal as a municipal service similar to water supply, sewage treatment, and fire protection, or is it a commercial function for the private sector to provide subject to public regulation similar to telephone service and cable TV? Perhaps it's a combination. It's common for private waste haulers to dump at municipally-owned and operated landfills, although, private landfills are not uncommon. For waste-to-energy facilities, the recent trend has been the private or commercial approach especially for the larger facilities. Some local officials feel the private sector can deal with the complexities and variables more effectively and expeditiously outside of the political arena. Others recognize that no contract with a private company can relieve the local government from the responsibility of solid waste disposal and associated environmental impacts, so it doesn't make sense to give up control.

How Much Can We Recycle? OK, but regardless of responsibility, authority, and public policy, markets for recyclables exist and are increasing, so what are the numbers? If we really commit to recycling and do the best we can, what would be the impact in solid waste management? EPA has suggested 25% as the national goal for recycling, and most states who have established a recycling objective have adopted this figure. Some believe objectives are intentionally optimistic by definition and therefore 25% will never be realized. Recycling advocates believe we will be surprised by the underestimated public participation and corresponding response by the marketplace for recyclables. Experts on both sides of the issue agree that recycling is still in its infancy and only time will tell. Once the public and media become aware of the environmental impacts and real costs of recycling, its popular support will probably wane somewhat. Most of the higher figures that are reported for selected communities are distorted by not including the waste from residents who chose not to participate in the curbside program. Or, more likely, they refer to the amount of material recovered with no indication of how much was actually reused. For instance, if a local recycling broker accepts recyclables to fulfill his agreement, but throws part or all of it in landfill because his market has temporarily 'softened,' it's still garbage but shows up on the monthly tonnage report as 'recycled.'

Even with a maximum commitment, it is hard to imagine that the diverse society of a spacious America could match the recycling effort of the crowded, highly disciplined, relative homogenous societies of Japan and West Germany. Table 1 compares the current percentages of the two nations' recycling with that in the U.S. The EPA 25% objective looks reasonable as a long range figure. Most experienced practitioners mention 15% as a more realistic figure for the remainder of this century. Incidentally, the widely reported 50% recycling figure for Japan has recently been clouded by a letter (6) from the Japanese National Institute for Environmental Studies to Dr. Walter Shaub, U.S. Conference of Mayors, Washington, D.C. The letter reports a national recycling figure in Japan of less than 10%.

If we had a national policy for recycling similar to PURPA for energy that guaranteed local governments a market for a certain amount of recovered materials, recycling would be on the front line of solid waste management along with waste-to-energy.

MYTH NO. 7: THE ENVIRONMENTAL IMPACTS ASSOCIATED WITH RECYCLING ARE VERY SMALL, ESPECIALLY WHEN COMPARED TO LANDFILLS AND COMBUSTION.

As mentioned earlier, the role of recycling in solid waste management is still emerging, and a lot of the specifics regarding cost effectiveness and environmental impacts are yet to be addressed. However, recycling is clearly not free of environmental problems and depending on the process, recycling may encompass adverse environmental trade-offs when compared to other solid waste disposal strategies.

As an example, newsprint is usually regarded as one of the most recyclable constituents in MSW. We have a pretty good idea of the environmental impacts associated with the combustion of newspapers because of the many tests which have been conducted on MWC stack and ash emissions. But, what are the environmental impacts associated with landfilling or recycling newsprint? A recent series of analyses(7) was conducted to review the toxicity of the newsprint of a typical U.S. newspaper with a circulation of 641,000 per issue. The annual output of this paper including daily and Sunday editions with all supplements was about 170,000 lbs.

Organics(7)	Lbs/yr
Methane Chloride	28.46
Acetone	134.2
Toluene	278
Ethylbenzene	71.4
Xylenes	348
Hexachlorocyclohexane	7.82
Phenols	360
TOTAL	**1,227.88**

Additionally, the total concentration of dioxin/furan in the newspaper was 248.49 parts per trillion (ppt). This translates to 11.158 ppt of TCDD toxic equivalent basis (U.S. EPA method). This amounts to more than 1 gram of TCDD equivalents. In comparison, a modern MWC facility which combusts

170,000 tons per year of MSW emits about 20 times less dioxin/furan to the environment.

MYTH NO. 8: DIOXIN CREATED BY THE BURNING OF MSW IS A KNOWN THREAT TO HUMAN HEALTH AND MWC FACILITIES POSE AN UNACCEPTABLE PUBLIC HEALTH RISK.

In spite of public perceptions, the impact of dioxin on human health has not been scientifically established. There are still no reported human deaths related to dioxin exposure. Over 30,000 residents in Sevesto, Italy were exposed to dioxin and subsequent medical examinations conducted by the AMA have indicated no long term health effects.(8) The American Society of Mechanical Engineers has a library which contains about 1400 publications on dioxins and furans. One of the best publications on the topic, particularly for the lay investigator, appeared in the February, 1986 issue of Scientific American, entitled: "Dioxin."(9) The subtitle of the article is: "Concern that this material is harmful to health or the environment may be misplaced. Although it is toxic to certain animals, evidence is lacking that it has any serious long term effect on human beings." The closing paragraph: "Two years ago a conference on dioxin at Michigan Sate concluded that the TCDD [most threatening form of dioxin] case is relatively less important than a number of other issues and that the nation's limited scientific resources should be devoted to the issues posing a greater threat. On the basis of the evidence turned up so far, the conclusion is still valid."

Dioxin is ubiquitous. It has been detected worldwide even in areas remote to industry. Its molecular structure comprises carbon, hydrogen, oxygen, and chlorine, four very common elements. It is created in various manufacturing processes and is contained in many consumer products, especially paper products. It is contained in coffee drip filters. One of the processes that can produce dioxin is combustion and incomplete combustion has the potential to produce more dioxin than complete, well controlled, combustion. Not surprisingly, forest fires, residential fireplaces, wood stoves, and open burning are examples of incomplete combustion and therefore, along with vehicle exhaust are potential sources of dioxins and furans. Smoke is an obvious indication of incomplete combustion, but a more meaningful indicator of the "completeness" of combustion is carbon monoxide (CO) emissions.

The automatic combustion controls in modern MWC facilities include continuous monitoring of CO and corresponding regulation of the combustion variables (fuel, air, temperature, and turbulence) to minimize CO. Testing confirms that dioxin/furan emissions from the furnace are minimized when CO is minimized. Naturally, the average citizen in a community about to implement a WTE facility is not satisfied with words like 'minimum' particularly when it applies to a chemical that has been reported as the "most toxic." Regardless of the fact that existing sources contribute more dioxin to his environment than the proposed WTE facility, he wants to know if it is safe. Just what is the health risk? To him?

Normally, health risk assessments are performed as a mathematical model of a proposed new source of pollution. Environmental scientists use an EPA computer model to predict the dispersion of the emissions from the stack, and maximum concentrations of various pollutants at ground level to indicate relative exposure of the population. The term 'maximum concentration' is used because the actual concentrations vary continuously with wind direction and velocity. In order to remove some of the variables associated with demographics, the EPA modeling procedures assume a fictitious person is continually exposed to the maximum concentrations of the pollutants. He is know as the MEI (maximally exposed individual) and he never checks out; but rather magically moves around night and day staying precisely at the point of maximum exposure for 70 years! This obviously over-predicts the public health risk but most everyone agrees the increased factor of safety is prudent. Even so, these models consistently predict low health risks to the MEI, which are quantified as the increased odds of contracting cancer due to the exposure. Of course, there is always the criticism that mathematical models are unreliable in terms of predicting the ground level concentrations.

At least one health risk assessment has been performed based on actual air samples at an operating facility. Table 8 is taken from the health risk assessment performed for the Pinellas County Waste-to-Energy facility and indicates the health risk for the MEI based on exposure to the air emissions. Interestingly, it compares this risk figure to the health risks of other common facilities, and the difference of the values is dramatic. A pint of milk per day is 700 times more risky than the air pollutions from a WTE facility?

MYTH NO. 9: THE FUNDAMENTAL QUESTION FACING A COMMUNITY IS: "DO WE BURN OR DO WE RECYCLE" BECAUSE RECYCLING AND COMBUSTION ARE INCOMPATIBLE.

This is also referred to as the 'burning issue' and it's another example of sleight-of-hand rhetoric designed to be 'good copy' headline material. It's catchy, simple, appeals directly to the common sense of someone not familiar with the real issues, and creates controversy by suggesting to the reader that he must first decide this fundamental question before addressing related questions. It is also damaging to local government dealing with solid waste management because it introduces the uninformed citizen to this issue with incorrect fundamental logic that is misleading and counter-productive public education. Logically, it is similar to the grade school trick question: "Do you walk to school or bring your lunch?" The correct logic is a series of questions that follow the 4-part integrated approach: 1) Do we as a community commit to waste reduction and recycling to reduce the amount of solid waste that must be disposed? 2) In regard to the disposal of what's left, do we bury untreated waste that is chemically and biologically active residue/ash in the landfill. Asking the public to choose between burning and recycling suggests there are two means of solid waste disposal available to the local government and it's an either/or situation.

TABLE 8. HEALTH RISKS (CANCER) ASSOCIATED WITH COMMONPLACE ACTIVITIES(10)

ACTIVITY (CAUSE OF CANCER)	LIFETIME RISK (CHANCES IN A MILLION)	RATIO OF RISK TO W-T-E RRF RISK
Cosmic Rays - Radiation Risk		
One transcontinental flight/year	35	29.4
Airline pilot - 50 hr/mo at 35,000 feet	3,500	2,941.1
Frequent airline passenger	1,100	909.0
Living in Denver compared to New York	700	588.0
Other Radiation Risks		
Living in brick building compared to wood (due to release of radon from bricks)	350	294.0
Natural radiation background at sea level	1,400	1,162.0
One chest x-ray every five years beginning at age 20 (ten total)	10	8.33
Cancer Risks in Eating and Drinking		
One diet soda/day (saccharin)	700	588.0
Four tablespoons peanut butter/day (aflatoxin)	2,800	2,325.0
One pint of milk per day (aflatoxin)	700	588.0
Miami or New Orleans drinking water (chloroform)	84	71.0
Half-pound charcoal-broiled steak one per week (benzo(a)pyrene) (cancer risk only)	28	23.3
Tobacco		
Smoker (cancer only)	84,000	71,428.0
Smoker (all effects)	210,000	166,666.0
Person in room with smoker	700	588.0
Air Pollution		
Average over U.S., all causes	17,500	15,151.0
Cancer risk only	1,050	909.0
Risk of Being Murdered (U.S.)	7,500	6,250.0
Modern Waste-to-Energy Facility	1.2	1.0

One is high tech, threatening, and expensive, the other low tech, non-threatening and seemingly inexpensive. It is easy to predict which the public would choose. Unfortunately, the public has been misled and the effectiveness of local government officials trying to solve the problem has been compromised.

WTE facilities recycle the heat in the various components in the waste stream. For some reason, the public perception is that recycling of heat does not count as real recycling, and the economics of WTE projects is comprised by the recovery of components that contain heat such as paper and plastics. The fact is that MWC and WTE facilities are benefitted by the removal of most recyclables. Certainly there is nothing to gain by trying to burn glass, metals, leaves and grass. Paper is most often recycled in the form of newsprint and corrugated boxes both of which contain heat, but do not account for enough heat in the waste stream to be significant. Plastic has a lot of heat, but the plastic products that are recovered by recycling operations are for the most part beverage bottles which only make up about .25%(2) of the plastic fraction which makes up about 7.2%(2) of the waste stream.(2)

The heating value of MSW is steadily rising due to compositional changes. Just a few years ago, the average heating value was accepted to be about 4500 BTU/lb. Today it is probably over 5000 BTU/lb. It has been reported as high as 6200 BTU/lb. at one existing facility.

As the heating value increases, the throughput or amount of MSW processed by a WTE facility, decreases due to the fact that a furnace/boiler is designed for heat throughput rather than fuel quantity throughput. As the heating value increases, existing facilities may not be able to process the amount of MSW that was originally anticipated which compromises the economic performance of the project. Therefore, recycling is regarded by the waste-to-energy industry as a welcome complement to solid waste management and disposal.

MYTH NO. 10: RDF COMBUSTION IS BETTER THAN MASS BURN COMBUSTION BECAUSE IT CREATES LESS ENVIRONMENTAL IMPACT AND INHERENTLY INCLUDES RECYCLING.

As mentioned in the discussion of Myth No. 3, emissions of air pollutants from modern MWC facilities comprise a 3-part strategy: 1) the furnace and combustion controls are designed to minimize the formation and emissions of organic pollutants including dioxins; 2) acid gas scrubbers remove most of the sulfur dioxide and hydrogen chloride gases from the furnace exhaust. Current technology provides for about 70% removal of SO_2 and 90% removal of HCL; 3) finally, particulate, or dust, collectors remove 99+% of the particles from the exhaust gas prior to stack emission.

The acid gas and particulate removal are external air pollution control devices that can be applied to the furnace exhausts from any boiler burning any fuel. The furnace and combustion control technology for organics, however, have been specifically developed for MWC units.

Because the physical and compositional characteristics of RDF are more consistent than MSW, and because RDF results from MSW processing, there is a perception based on misleading information that RDF is "cleaner;" and therefore burns better producing less pollution. As indicated above, however, less pollution must mean organics, or dioxins, because that is the only part of the pollution control system that is different for different fuels. Testing to date indicates just the opposite. EPA conducted a series of tests on the emission from two RDF facilities and two mass burn facilities. The dioxin/furan emissions from the RDF furnaces were several times higher than those from the mass burn facilities.(11)(12) probably due to the increased flyash emissions inherent to RDF combustion. This does not mean RDF combustion cannot be as complete as mass burn combustion, it just means RDF is not a cleaner burning fuel or a better combustion technology.

The decision between RDF and mass burn combustion is not based on environmental factors, nor is it based on recycling, because recycling is not a function of technology or the systems used for solid waste management. As mentioned earlier, recycling is a management strategy and the success of recycling is primarily dependent on markets for the recyclables. Determining the best means of recovering the recyclables from the waste stream is a separate question.

There are two reasons to use RDF technology as opposed to mass burn technology in a WTE facility. 1) If it is more cost effective, that is, if the cost of processing the waste is at least offset by the cost reductions allowed by burning the waste in smaller more physically consistent pieces. 2) If the application is to produce and sell fuel rather than steam or electricity.

The flip side of the built-in recycling aspect to RDF is the disposal factor. The main idea of any WTE facility is to provide disposal of MSW. Most of the technology arguments hinge on which one is more cost effective or environmentally better, etc. However, keeping your eye on the disposal target, RDF inherently rejects 40% to 60% of the processible waste stream to recycling operations or landfill, whereas with mass burn, 100% of the processible waste stream is exposed to the furnace environment reducing the volume of waste stream by 80% to 90%. The processible waste stream is that being received at the plant after curbside recycling and special collections have removed reusable and compostable components. More disposal bang for the buck.

PUBLIC INTEREST NOT SERVED BY RHETORIC

The dynamics of social debate in a free society allow the opposition party the advantage of unaccountability. That is, anyone or group can oppose anything on virtually any grounds. However, the greater technical, legal, and financial community trying to promote solutions for local government is required to substantiate their positions and initiatives both within the professional community in the form of peer review, and without in the form of public scrutiny.

Most experienced solid waste management professionals and practitioners are not sure of WHO is trying to effect WHAT social change at any given time, so their efforts to deal with solid waste issues are often confounded by the hidden agendas of the volunteer organizations opposing any particular aspect such as MWC. The practitioners usually work for the local government in some fashion and therefore, focus on the technical, environmental, and economic aspects of the problem, which is: "What is the best way to dispose of the trash that the public expects to have picked up at the curb twice a week?" In developing programs that address this problem, the local governments who are responsible to ALL citizens must be able to establish sound policies that employ any and all strategies that are proven to be safe and cost effective. One of these strategies is modern municipal waste combustion in the form of waste-to-energy or resource recovery facilities.

The current clamor over the acceptance of MWC, particularly the mass burn technology, is counter-productive to the development of feasible solid waste management and does not serve the public interest. The clamor, however, could be transformed into debate that would be beneficial to all perspectives if all parties involved would participate in the national and international meetings (such as this conference) sponsored by EPA, U.S. Conference of Mayors, and the professional engineering societies. Debate, especially strong debate, in the public and professional arenas is the fuel for sound public policy so long as all parties are held accountable for their information and are prepared to be responsible for their positions. For instance, if someone or group promotes the idea to the public that we recycle our way to solid waste disposal, then they should indicate to the public HOW they will be accountable for its effectiveness and cost effectiveness.

Without this transformation from clamor to debate, the politics of garbage, like its predecessor, the politics of energy, will lead to the continued development of solid waste management strategies that do not offend anyone in the short term but also don't solve the problem in the long term. So long as we have the clamor, we will not be able to establish a national solid waste management and disposal policy that effectively deals with the overall problem.

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AN ANALYSIS AND DECISION MODEL FOR INTEGRATING ENERGY AND MATERIALS RECOVERY WITHIN A SOLID WASTE MANAGEMENT SYSTEM

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ABSTRACT

Integration of extensive source separation, intermediate materials processing and recycling with energy recovery and landfilling has been receiving considerable attention and is gaining increasing acceptance as an approach to solid waste management. The concept has arisen from the debate between recycling advocates and the waste-to-energy advocates, which was fueled by the perception that waste-to-energy and recycling are mutually exclusive and must compete for a common waste stream. However, the integrated approach to be described in this paper enables the development of source separation, materials recovery, and recycling so as to attain technical and economic compatibility between all components of the system. Component sizing and timing alternatives can be analyzed in an effort to maximize waste volume reduction and overall net energy yield, and minimize landfilling, environmental impacts and total system costs.

The paper will present a decision model which evaluates various alternatives and scenarios necessary to bring the integrated solid waste management plan into reality. This model assists decision makers by evaluating and refining capacity decisions with respect to the various components such as the energy recovery facility (ERF), materials recovery center (MRC), composting, construction and demolition debris recovery and landfill needs on a technical and economic basis. A case study will be presented where this model has been utilized.

The technical evaluation matches local waste characteristics with the most appropriate waste management techniques. The economic evaluation will compare the system costs for all the alternatives analyzed, and includes financial and economic criteria and life cycle cost/benefit analysis. The model will also predict solid waste higher heating values net of source separated materials, potential for waste-to-energy facility downsizing or future expansion, and effects on facility operating economics.

BASIC PROCESSES FOR CLEANING FLUE GASES FROM WASTE INCINERATION PLANTS

ASSESSMENT IN CONSIDERATION OF THE GERMAN CLEAN AIR CODE TA LUFT 1986

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ABSTRACT

Dry, quasi-dry and wet processes with wet and dry residue discharge are applied for cleaning the waste gases from municipal and hazardous waste incineration plants. While purely dry processes may sometimes fail to reach the stipulated collection efficiencies for gaseous mercury, SO_2 and HCl can be removed to a sufficient extent, if an increased amount of adsorbent is used. Quasi-dry processes guarantee that the emission requirements stipulated in TA Luft 1986 will be met with a reasonable amount of expenditure in the majority of cases. The application of wet processes with upstream particulate collector permit even better emission values to be achieved particularly as far as heavy metals are concerned.

The decision as to which process represents the optimum solution depends on a large number of factors, such as location, fuel and dumping conditions. This paper deals with plant concepts corresponding to the present state of the art, and concentrates on special aspects of component combination and integration into noxious gas removal systems. All the processes dealt with are being applied in industrial plants.

In general, it can be said that gas cleaning technologies have reached a standard which ensures that the values stipulated in TA Luft 1986, for both municipal and hazardous waste incineration plants, cannot only be safely met but even underrun.

Introduction

The stipulations of TA-Luft 1986 relative to waste incineration plants, and the critical situation in the field of residue disposal led to an interesting variety of plant components which can be combined to form integrated systems for dust collection and noxious gas removal.

The following has to be taken into consideration for the selection of suitable flue gas cleaning processes:

- the emission standards, particularly for dusts, noxious gases and heavy metals,
- the possibility of using existing plant sections, such as electrostatic precipitators, waste water systems, stacks
- the disposal of residues (dry/wet; dumping regulations and dumping cost; possibility of re-using or of recycling the residues, capacity of the receiver)

In view of the above it is impossible to give general preference to a specificic gas cleaning process. Therefore, the optinum solution to the problems of the individual plants has to be found.

Raw Gas Mass Flow and Pollutant Data

For the design of the flue gas cleaning system it is necessary to know the mass flows entering and leaving the refsue incineration plant.

On drastic simplification and limitation to the main mass flows, we obtain the values shown in table 1 .

Table 1. Raw gas mass flow and pollutant data

Primary mass flows per ton of domestic waste

	Unit	Varying between	Mean values in the raw gas
Flue gas	m^3	5000-5500	5250
Water vapour	m^3	700-1100	900
Slag	kg	200-400	300
Fly ash	kg	10-30	20

Noxious gas per ton of domestic waste

	Untreated domestic waste in kg	Raw gas in kg	Mean raw gas loads in mg/m^3
HCl	6-8	5,0-7	1100
SO_2	2-6	1,5-2,5	400
HF	0,05-0,2	0,02-0,08	10
NO_x	1,5-2,5	1,5-2,5	350

Volumes and loads indicated are based on dry flue gas at STP (1013 mbar, 273 K and 11 % O_2)

Taking as a basis the mean emission values shown in table 1 and assuming that the pollutants SO_2, HCl and HF are removed in accordance with the values stipulated in TA Luft, the quantities of residues (fly ash plus reaction products - absorbent lime) collected in the downstream flue gas cleaning system will be 31.5 kg/h for a stoichiometric factor of 1, 35 kg/h for a stoichiometric factor of 1.5, and 38.5 kg/h for a stoichiometric factor of 2. These values may be increased due to process-depending and operation-related $CaCO_3$ formation.

Heavy metals, which reach the combustion process in remarkable concentrations, leave it either via the slag or the raw gas. For the gas cleaning process it is of particular importance to know which chemical compounds have to be reckoned with and whether they will be present in the solid or the gas phase.

Test results of various parties [1 to 3] completed by our own test results have been compiled in table 2, which shows the percentages of heavy metal emissions to be reckoned with in the raw gas. The rest remains in the slag in the form of water-insoluble hydroxides or carbonates.

The heavy metals present in the raw gas as chlorides, sulfates, oxides etc., pass through various temperature stages, condense and get adsorbed, mostly onto the fly ash surface.

The question as to the heavy metals or heavy metal compounds that reach the gas cleaning system in their gaseous phase is answered by fig. 1. With temperatures of less than 200 °C, it is only the vapour pressure of mercury that is worth mentioning. Tests carried out showed that this heavy metal would normally occur as bivalent halide in waste incineration plants [4].

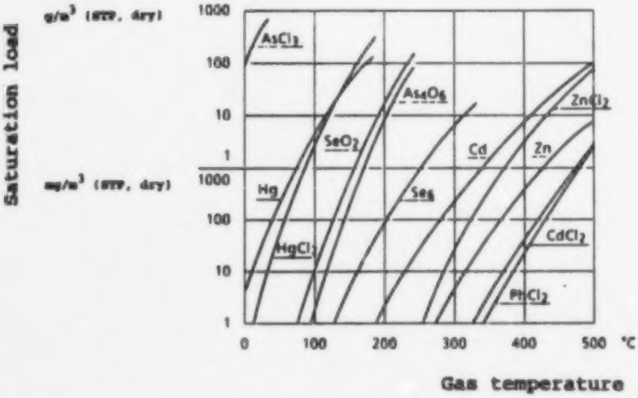


Figure 1. Saturation loads of gaseous heavy metals and heavy metal coumpounds

With domestic waste incineration, mercury concentrations will normally vary between 0.3 to 0.6 mg/m3, with industrial waste the values may be considerably higher. Removal of gaseous mercury has become a very decisive factor for the selection of the gas cleaning process.

Table 2. Heavy metal content of the untreated waste and corresponding raw gas emissions in %

	Heavy metal load of untreated waste in g/t	Raw gas in %
Mercury	2-4	> 95
Cadmium	1-10	75-90
Arsenic	0,2-4	75
Zinc	1000-2000	27-40
Lead	400-1200	30-35
Copper	400-600	< 10
Nickel	15-85	< 10
Chromium	50-250	< 10

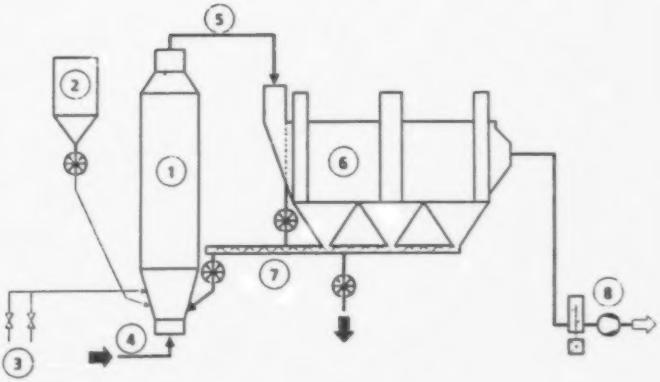
Process Selection

Three different basic processes have become generally accepted in the past. To meet the requirements of the latest version of TA Luft, these processes, that is dry, semi-dry and wet waste gas cleaning, have been constantly modified. The process variants, their specific features and their results shall be studied in this paper on the basis of the experience gathered in industrial plants.

Dry Waste Gas Cleaning

As for dry waste gas cleaning, the principle of the circulating fluid bed has been applied in several industrial plants.

The dust and noxious gas laden flue gas produced by the combustion process is passed through a fluid bed reactor where it gets into intimate contact with the fine grained adsorbent, which can be fed into the reactor either in the form of a powder or of a suspension. Optimum adsorption conditions are achieved in the reactor by the high flow and relative velocities as well as by the uniform axial gas/solids distribution and the high bed density (fig.2).



1) Venturi reactor
2) Absorbent
3) Addition of water
4) flue gas route
5) Flue gas route
6) Elektrostatic precipitator
7) Circulation
8) Fan

Figure 2. Dry waste gas cleaning in circulating fluid bed

The solids leave with the flue gas via the reactor top and pass to the precipitator downstream of the reactor. The bulk of the solids is removed from the gas flow in a louver-type pre-separator and recycled to the reactor. Constant recirculation permits solids retention times of up to half an hour.

The typical process characteristics of the circulating fluid bed are:

- intensive noxious gas/solids mass transfer
- even gas/solids distribution across the entire reactor height and cross section
- long solids retention times, therefore efficient lime utilization at low stoichiometrical values
- simple plant design without sophisticated moving parts.

The process was applied on an industrial scale to a cogeneration plant based on waste combustion. This power station, which is located in Bavaria, takes care of the waste disposal of several rural districts and towns. In this power station three parallel units having a waste throughput of 18 t/h are followed by a gas cleaning system operating according to the principle described above. The start-up took place in 1982, that means, the clean air requirements of TA Luft applicable at that time had to be met.

The values measured for two different temperatures in 1984 are shown in table 3. It should be mentioned that lower temperatures can be adjusted by conditioning with water. The results clearly confirm the excellent HCl and SO_2 adsorption of circulating fluid beds, which can be considerably improved with low flue gas temperatures. However, with 100% dry operation, it is not possible to meet all the Hg limits stipulated in TA Luft 1986. A reduction of the temperature to values clearly under 160°C is inevitable for this purpose.

Table 3. MSW - Dry gas cleaning

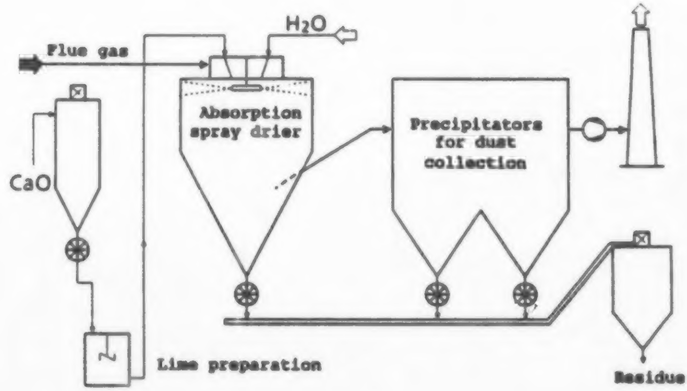
		Raw gas	Clean gas	Clean gas
Gas temperature	° C	240-280	240-280	160
Dust load	g/m^3 (STP,dry)	200-600	0,04-0,06	< 15
HCl	mg/m^3(STP,dry)	700-1500	< 30	< 15
HF	mg/m^3(STP,dry)	≈ 15	< 1	< 1
SO_2	mg/m^3(STP,dry)	400-600	< 100	< 50
Heavy metals Pb,Cd,Hg,As (Dust) Collection efficiency	%		99	> 99
Hg (vapour) Collection efficiency	%		37	46

Quasi-dry Waste Gas Cleaning - Spray Absorption

Spray absorption was developed on the basis of the spray drying process, a technology that goes back to patent applications filed by Metallgesellschaft in the early twenties and applied by Lurgi Wärme as early as 1922.

Spray absorption has been frequently applied for treating flue gases from waste incineration plants and power stations in recent years and it will remain an interesting technology for future projects, too.

In the spray absorption process, the absorption agent is fed onto a fast rotating disk located in the spray absorber. Under the influence of the centrifugal force, the absorption agent is forced to the outer rim of the disk where it is atomized by the high shearing forces (fig. 3).



H_2O
Flue gas
Absorption spray drier
Precipitators for dust collection
CaO
Lime preparation
Residue

Figure 3. Spray absorption

The liquid mist forced to the periphery in a fast radial and spiral movement is brought into intimate contact with the flue gases. Gas absorption takes place at a high reaction velocity, first in the liquid and later in the dry phase with simultaneous cooling of the flue gases by the evaporating water. While part of the dry residue is discharged from the reactor cone, the remainder is collected in a downstream particulate collector.

To ensure proper operation from the process point of view as well as from the equipment point of view, the following has to be taken into account in spray absorber design:

- The flue gas must enter the reactor space according to a defined flow pattern. An even flow through the reactor must be ensured even with varying load conditions.
- The flow route must be such as to avoid deposits on the inner surfaces of the absorber.
- To achieve a proper mass transfer, it is necessary to adjust high relative velocities between droplets and flue gases.
- The spraying equipment including the supply connections has to be of a design permitting replacement within a few minutes.
- Design and material of the atomizing disk should guarantee that corrosion, erosion and deposits within the disk area are either prevented or reduced to a minimum to achieve maximum availability.

Both electrostatic precipitators and bag filters are used for dust collection downstream of the absorber. Bag filters have the advantage of adsorbing residual gaseous pollutants onto the dust deposited on the bag surface. However, the low temperature limit of the flue gas must not be underrun since the crystal water added onto the $CaCl_2$ may lead to fabric clogging. Moreover, special measures have to be taken to prevent the temperature from dropping below the dew point during start-up and shut-down procedures. Due to the reduced danger of deposit formation, precipitators, on the other hand, permit a further reduction of the flue gas temperature and thus a further increase in the adsorption efficiency of the spray reactor.

The stoichiometric factors achievable in practice depend on several parameters, the most important being the flue gas temperature, the collection efficiency and the reduction of the pollutant concentrations to specified for the clean gas side. Extremely low clean gas values make the reagent consumption and at the same time the free lime in the residue rise disproportionately. This will particularly be so whenever SO_2 values are required which are far below the values stipulated in TA-Luft. In this case the SO_2 concentration in the cleaned gas has to be used as reference component for process control.

In general, the stipulations of article 5 of the German federal law on the prevention of immissions (BImSchG) are applicable to residues from waste incineration plants. These stipulations concentrate on such aspects as the reduction of soluble salts and the minimization of residues by concentration. The present practice, however, is to dump the residues on controlled landfills for hazardous waste. So the next step must be a considerable improvement of the elution properties of the residues so that less exacting dumping requirements become applicable. Moreover, there is a good chance of producing a stable product which, if sufficiently resistant, will permit application in underground and overground construction.

On the whole, three ways of processing are being studied:

- Solidification of residues by adding large quantities of binding agents. However, no definite test results as to the long-term immobilization of chlorides and heavy metals are available so far.
- Alcaline scrubbing. The chloride loads can easily be removed, heavy metals dissolve only to a small degree. The residue may be solidified by adding minor quantities of binding agents. The elution properties of heavy metals are considerably improved.
- Thermal treatment of the residue with a view to achieving a product which can be marketed as raw material or intermediate product for the building industry. The drawback is that this market will probably decrease in the future and that more substitute products will certainly appear on the market.

Wet Waste Gas Cleaning

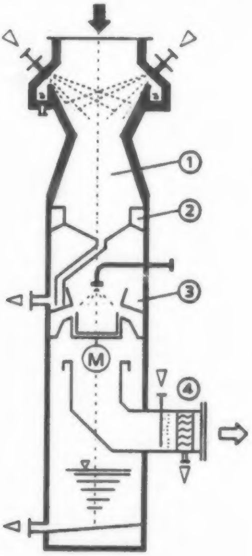
Wet processes for flue gas cleaning are applied whenever the following has to be taken into account:

- minimization of the mass of the residue since stoichiometric factors of approx. 1 can be achieved,
- possibility of discharging chlorides to the receiver,
- very high content of gaseous heavy metals in the raw gas,
- stringent requirements as to the heavy metal content in the clean gas (<0.1 mg/m^3),
- existence of vital plant sections, such as dust collector, waste water treatment plant, stack etc.

Wet processes are classified according to whether waste water is produced or not. With the aid of examples of plants of different design for various applications of incineration of both domestic and chemical waste, various designs for wet flue gas cleaning systems shall be dealt with here.

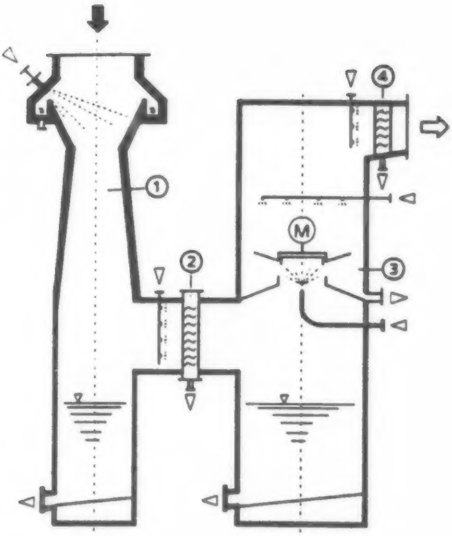
The vital part of wet-type gas cleaning systems are scrubbers for gas cooling, solids separation and adsorption of gaseous pollutants. While in the past scrubbers were primarily used for dust collection, they are nowadays employed as fine cleaning stage in waste incineration plants, for the removal of SO_2, HCl, HF and gaseous heavy metals.

The radial flow scrubber is a device that has proven very successful in practice. It is used in a diversity of designs depending on the application and the prevailing operating conditions. It may be combined with a separate cooling tower or a venturi scrubber or equipped with a venturi scrubbing stage at the scrubber inlet. Figures 4 and 5 show two basic designs. The first is characterized by small space requirements which make it particulary suitable for retrofittings. The second design, although requiring more space, offers the advantage of a reduced pressure drop and of a clear separation of the individual circuits.



1) Saturator venturi/ 1st scrubbing stage
2) Centrifugal droplet separator
3) Radial flow scrubber/ 2nd scrubbing stage
4) Lamellar droplet separator

Figure 4. Saturator venturi with radial flow scrubber (space saving design)



1) Saturator venturi/ 1st scrubbing stage
2) Lamellar droplet separator
3) Radial flow scrubber/2nd scrubbing stage
4) Lamellar droplet separator

Figure 5. Saturator venturi with radial flow scrubber (separate arragement)

Processes involving Waste Water--

Incineration Plants for Municipal Waste--

Downstream of waste incineration plants we frequently find hot gas precipitators so that the treatment stage for gaseous pollutants which has to be added may have a low residual dust collection efficiency. The above mentioned scrubber and the arrangement shown in fig. 6 suggest themselves for this purpose.

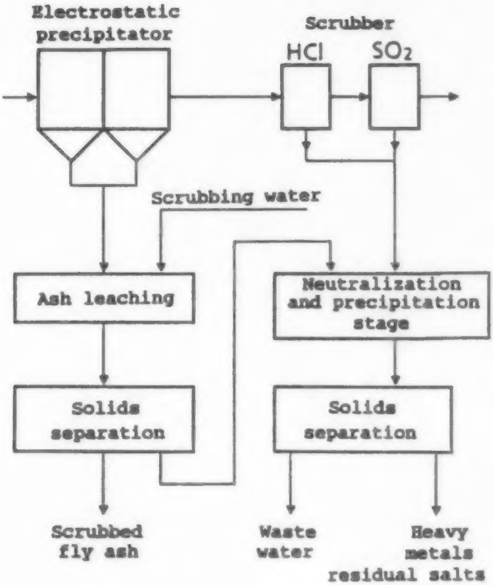

Electrostatic precipitator
Scrubber
HCl
SO2
Scrubbing water
Ash leaching
Neutralization and precipitation stage
Solids separation
Solids separation
Scrubbed fly ash
Waste water
Heavy metals residual salts

Figure 6. Flue gas cleaning with residue treatment

The flue gases coming from the waste incineration plant are pre-cleaned in precipitators before they enter the radial flow scrubber. After being cooled down to dew point temperature, they reach the first scrubbing stage. In this scrubbing stage, which is operated at a low pH, HCL and HF are removed to a large extent. SO_2 is removed in the second scrubbing stage where the environment is alcaline. To avoid concentration of the recycled scrubbing fluid in the scrubber, part of the fluid is continuously withdrawn, neutralized outside the system by means of milk of lime and conducted to a waste water treatment plant for further processing.

The withdrawn scrubbing fluid is processed in a waste water treatment plant together with other effluents to allow discharge into the receiving water. The treatment comprises neutralization with milk of lime, which is followed by precipitation. The bulk of the sulfates and heavy metals is converted into water-insoluble compounds in this process step and separated from the waste water. In the subsequent precipitation and settling stage, specific precipitation agents for mercury and other heavy metals are added to remove these pollutants down to the residual values tolerated by the law.

In the waste water treatment plant, the pollutants are removed from the effluents in the form of excess sludge, which has to be thickened and then dewatered by means of a filter press.

Apart from the waste water, the fly ash collected in the precipitators is subjected to a treatment in order to remove the water-soluble salts and part of the heavy metals. This is done by leaching the ash with scrubbing acid in a mixing tank and by dewatering it in a belt filter before final cleaning.

The outstanding feature of the wet flue-gas cleaning process is its residue treatment method which provides for the residues to be obtained separately as :

- fly ash in its leached form permiting further use
- filter sludge consisting of residual dust, sulfates and heavy metals in concentrated form - the actual hazardous product to be dumped
- cleaned, chloride-laden effluent.

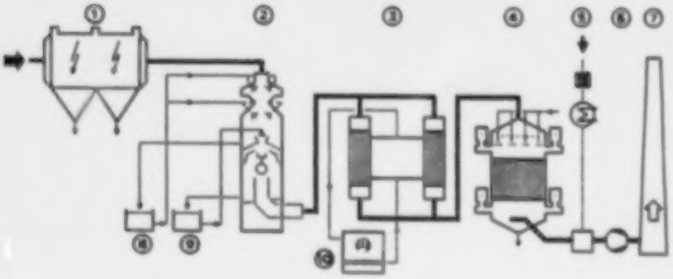
Depending on the proportions of the individual pollutants, the above method permits the remainder of the residues, which has to be taken to the hazardous waste dump, to be reduced to 15 - 30% of the original proportion. Some incineration plants are already being retrofitted for operation acc. to this process.

Incineration plant for chemical waste--

Cleaning of waste gases produced on incineration of chemical waste mostly focusses on aerosol removal. Wet precipitators are arranged downstream of radial flow scrubbers for this purpose as they offer a good but hitherto rarely used way for removing aerosols and heavy metals from off gases of waste incineration plants.

Wet-type electrostatic gas cleaning has for a long time been successfully applied in a number of fields where industrial or waste gases have to be cleaned down to very low residual pollutant contents. Wet precipitators achieve high efficiencies even on very fine particles with relatively low utility requirements.

If large proportions of gaseous mercury have to be reckoned with, the gas cleaning line between scrubber and wet precipitator mentioned above may be equipped with an indirectly cooled condensation stage. On lowering the flue gas temperature by 5° - 20°C below the saturation temperature, excellent conditions can be achieved for the absorption of gaseous heavy metals [5]. The first gas cleaning installation operating according to this process has been successfully in service since November 1987 (fig. 7).



1) Dry precipitator
2) Radial flow scrubber
3) Gas cooler
4) Wet precipitator
5) Flue gas reheating system
6) Induced draught fan
7) Stack
8) Acid scrubbing stage
9) Alkaline scrubbing stage
10) Water recooling system

Figure 7. Flue gas cleaning in hazardous waste incineration plant

Processes not involving Waste Water--

With Spray drier--

In a few important aspects, the advantages of wet gas cleaning may be linked to those of dry gas cleaning. This process variant works according to the principle shown in figure 8. The flue gases of the incinerator pass directly to the spray absorber where they are allowed to come into intimate contact with atomized hydrate of lime suspension for for extensive pollutant removal. HCl, HF and SO_2 react with the hydrate of lime forming a dry particulate residue which is discharged together with the fly ash. The particulate matter is collected either in a bag filter or an electrostatic precipitator.

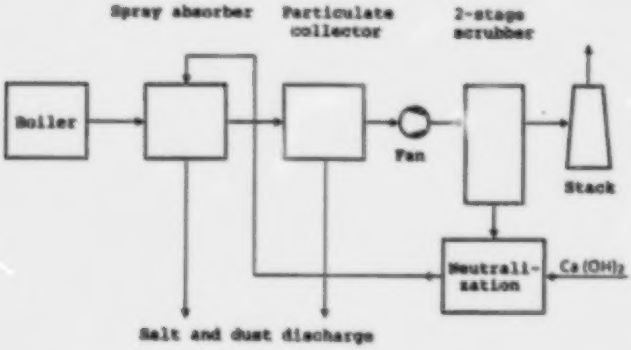

Spray absorber
Particulate collector
2-stage scrubber
Boiler
Fan
Stack
Neutralization
$Ca(OH)_2$
Salt and dust discharge

Figure 8. Flue gas cleaning with dry dust discharge

The next step is to pass the flue gas to a scrubber and to treat it, as described above, first in the acid stage and then in the alkaline stage to reduce the remaining content of gaseous pollutants, particulate matter and $HgCl_2$ to the values stipulated. If a high aerosol proportion has to be reckoned with, an electrostatic precipitator may be added downstream of the scrubber.

To prevent the recirculated scrubbing fliud from being concentrated in the radial flow scrubber, part of the fluid is continuously withdrawn from the scrubber circuit, neutralised and passed to the absorber for evaporation of the water content.

Apart from the normal dry reaction products such as $CaSO_4$, $CaSO_3$, $CaCl_2$, relatively small quantities of Na_2SO_3 are obtained in the absorber

The plants operating according to this process principle have several advantages, the most important being:

- the wet gas cleaning stage with its highly efficient removal of remaining gaseous heavy metals, HCl, HF and SO_2
- the absence of a waste water treatment system
- the low stoichiometric factors - in spite of low pollutant concentration
- the shifting of the separation process to the spray absorber and/or scrubber which increases the choice of absorbents that can be used.

With Evaporation Crystalliser--

Another possibility of applying the wet gas cleaning process without producing waste water is offered by the use of the evaporation crystalliser. The flue gas is treated in the same way as described above.

After waste water treatment, the clean effluent is passed to the multi-stage evaporation crystalliser for evaporation of the water and separation of the solids from the mother liquor to obtain a salt that permits industrial utilisation [6].

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OPERATING EXPERIENCE WITH THREE APC DESIGNS ON MUNICIPAL INCINERATORS

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Anthony Licata - Dravo, Long Beach
John Buschmann - Flakt, Inc.

ABSTRACT

Design features and operating experiences of four Air Pollution Control (APC) Systems utilizing three different designs, are reported. The first design includes two plants, (Bristol and Stanislaus), using a nozzle atomizer spray dryer and pulse jet cleaned baghouse. The second system, (Long Beach), utilizes a rotary atomizer spray dryer and reverse air cleaned baghouse. The third system, (Springfield), uses a heat exchanger for gas cooling, a dry lime injection system, and a pulse jet cleaned baghouse. Areas of interest are comparative capital and operating costs, maintenance requirements, and gas cleaning effectiveness. The systems are installed on incinerators of widely varying designs.

Reference herein to the specific companies and/or plants are for illustrative purposes and are not intended to constitute endorsement by these companies of the contents hereof or of any specific equipment.

SPRINGFIELD

The plant is located in the town of Agawam, Massachusetts, on the west bank of the Connecticut River. The facility consists of three combustion trains to burn municipal solid waste (MSW) and a turbine generator for electric power generation. Power is sold to the Western Massachusetts Electric Company (WMECO) and steam is sold to the Springfield Regional Wastewater Treatment Plant.

The Springfield Resource Recovery Facility serves the nine cities in Southwest Massachusetts; Springfield, Agawam, Long Meadow, East Long Meadow, Wilbraham, West Springfield, Heath, Ludlow, and Southwick.

The entire waste stream from these cities is burned by the plant. This waste averages 312 TPD, approximately 87% of the plant's rated capacity of 360 Tons Per Day (TPD). The contracted tipping fee is $24/ton. The project was developed by Vicon Recovery Systems. Fluor Daniel was the Engineer and Constructor. The plant is owned by a subsidiary of Fluor, Inc. Operator of the plant is Springfield Resource Recovery, Inc., also a subsidiary of Fluor, Inc. The total project value is approximately $34mm. The plant has been operating since May of 1988.

Each of the three parallel incinerator trains is designed to burn 120 tons per day of MSW having a higher heating value of 4500 BTU per pound. The annual guaranteed plant capacity is 116,270 tons of MSW, representing 88% capacity factor. A sketch showing an overview of the plant and the technology follows this description.

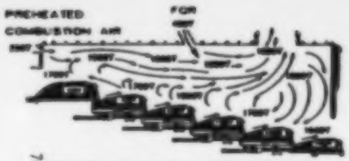
FIGURE 1

ENERCON "CONTROLLED TEMPERATURE" COMBUSTION TECHNOLOGY

ENERCON SYSTEMS, INC. - ELYRIA OHIO

ALL DATA IS ESTIMATED AND FOR REFERENCE ONLY

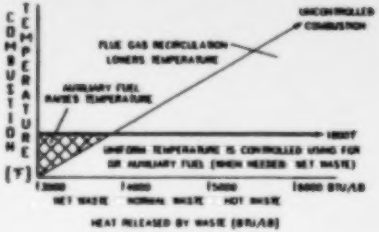
PRIMARY COMBUSTION CHAMBER


PREHEATED
COMBUSTION AIR
FGR

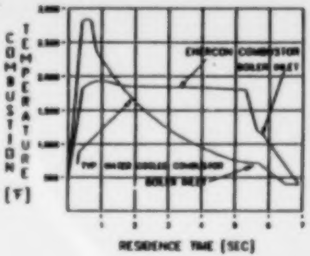
"REFRACTORY WALLS MAINTAIN UNIFORM TEMPERATURE
ACROSS AND WITH THE DIRECTION OF GAS FLOW

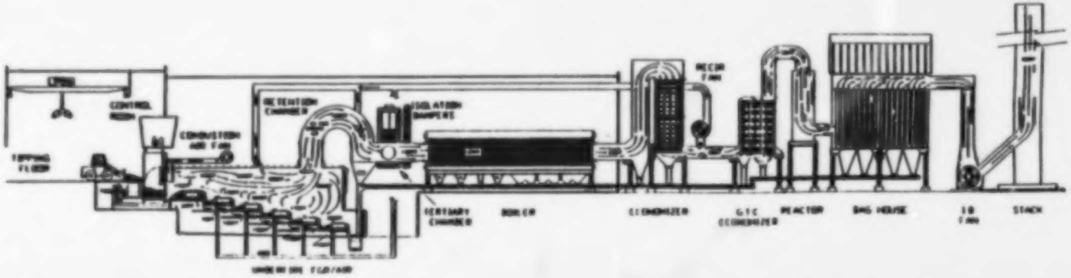
"APWALL" PLENUM PROVIDES COOLING FOR THE REFRACTORY WALLS
HEAT IS RECOVERED AND USED TO PREHEAT THE COMBUSTION AIR

CONTROL OF COMBUSTION TEMPERATURE
USING FLUE GAS RECIRCULATION (FGR) AND AUXILIARY FUEL



TIME Vs. COMBUSTION GAS TEMPERATURE




CONTROL ROOM
TIPPING FLOOR
COMBUSTION AIR FAN
RETENTION CHAMBER
ISOLATION DAMPERS
UNDERFIRE FGR/AIR
TERTIARY CHAMBER
BOILER
ECONOMIZER
RECIR FAN
G.T.C ECONOMIZER
REACTOR
BAG HOUSE
I.D. FAN
STACK

The municipal waste is received in trucks, which are weighed at the plant entrance. The waste is tipped in the receiving pit or on the tipping floor, where non-combustible material is separated from the acceptable waste using the overhead crane and front end loaders. The acceptable waste is dumped into the incinerator hoppers. The non-combustible material is stored in roll-off containers and transported to the landfill.

The tipping facilities and combustion systems are located inside a large building, permitting passage of air from the handling facilities into the incinerators, to minimize odor propagation.

The air for combustion is preheated both by recovery of heat in a plenum around the primary chamber, in a closed loop system for transfer ram cooling and in a steam heated exchanger.

The Primary combustion Chamber (PCC) consists of six (6) progressively lower refractory lined hearths resembling "steps". Solid waste is tumbled from "step-to-step" by hydraulically activated water cooled rams.

Combustion gas from each incinerator passes through a secondary combustion chamber and into a waste heat boiler which generates approximately 28,500 lb/hr of 650 psig 750°F steam at normal conditions.

The boilers share a common water treatment plant, condenser, cooling tower, and turbine.

Steam from the three boilers is combined for a total of approximately 85,500 lb/hr. Normal design is for an average of 4,700 lbs. per hour of steam sales to the waste water treatment plant, although capability is 0 to 14,700 lb/hr of sales steam. Recently, sales have been about 10,000 lb/hr. The remaining steam drives the turbine generator, which generates electricity for internal plant use and for sale to WMECO. Design generator capacity is 9.427 megawatts (MW) at 0.9 power factor of which about 6.3 MW will be available for sale.

Flue gas from the combustion trains is cooled, then scrubbed with dry calcium hydroxide for removal of acid gases. The cleaned flue gas then passes through a baghouse filter for particulate removal, then to the atmosphere through a 175 foot high stack.

Ash from the combustion process is quenched and cooled in ash troughs, then conveyed to a dumpster where it is transported to the Bondi Island landfill for disposal.

The facilities for handling MSW include both a storage pit and a tipping floor. A crane moves waste from the pit to the floor where front-end loaders sort the waste and remove unacceptable materials before loading waste into the feed hoppers. Waste can be dumped into the pit or directly on the floor if, for example, the crane is down for maintenance. Only one crane is provided since this design avoids the need for a spare.

The facility is controlled from a central control room. A Distributed Control System (DCS) collects plant operating information and displays it on Cathode Ray Tube (CRT) screens. The control room operator controls much of the plant equipment directly. Operating data logs are printed from the DCS and data necessary to retain long term is archived in the system.

The combustion/heat recovery system is sized to permit modular assembly and construction. Various incinerator modules are assembled on-site to minimize field erection while the boiler and economizers are shipped as modules for setting.

The surface condenser is designed to accept automatic by-passing of the turbine generator. High pressure and temperature steam is de-superheated and depressurized and condensed in the surface condenser when the turbine generator trips.

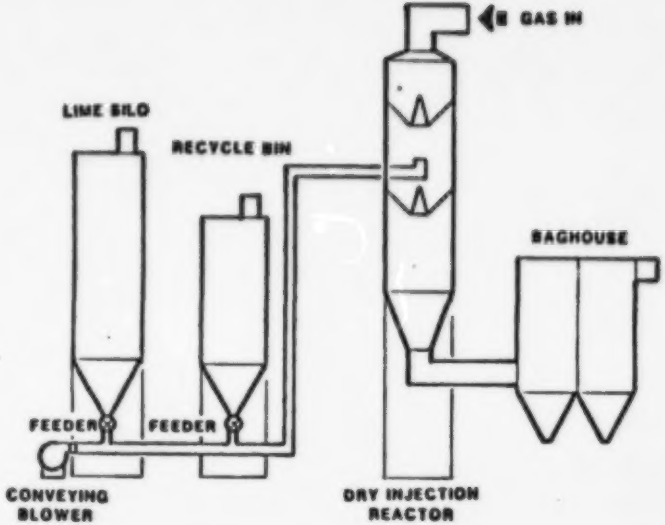
Flue gas leaving the economizer splits into two streams. The net production of flue gas flows to the air pollution control system while a recycle stream is returned for use in the incinerator to assist in controlling combustion and boiler entrance temperatures. The control of combustion temperatures reduces generation of nitrogen oxides.

Following the primary combustion chamber a retention chamber provides time for completion of combustion reactions. In combination with the control of temperature through flue gas recirculation, the retention chamber provides very low carbon monoxide and volatile organic compound concentrations in the flue gas. The concentrations of dioxins and furans are very low in the stack gases.

The flue gases leaving the economizer are further cooled in a gas temperature control heat exchanger before entering the air pollution clean-up system. The gas temperature control economizer both recovers additional energy and provides improved pollutant removal. Energy recovery is accomplished by circulating water through the gas temperature control economizer where it is heated by the flue gas and then through an exchanger where the heat recovered is used to preheat the condensate returning from the surface condenser to the deaerator.

The plant has 26 operators, 13 maintenance men, and 5 administrative staff.

The flue gas cleaning system utilizes dry injection of lime for acid gas control followed by a pulse jet cleaned baghouse for particulate control. The equipment is illustrated in Figure 2. After being cooled in a secondary economizer, the incinerator flue gas enters a dry absorption reactor (5'6" diameter by 17' straight side height) where it is mixed with dry, finely powdered absorbent, in this case, hydrated lime $Ca(OH)_2$, to remove acid gases. The absorbent is pneumatically injected through an abrasion resistant nozzle designed for proper distribution over the reactor area.


GAS IN
LIME SILO
RECYCLE BIN
BAGHOUSE
FEEDER
FEEDER
CONVEYING BLOWER
DRY INJECTION REACTOR

DRY ABSORPTION SYSTEM (DAS)

FIGURE 2

A chemical reaction binds the acidic substances present in the gas, primarily hydrogen chloride (HCl), hydrogen fluoride (HF), and sulphur dioxide (SO_2), to the lime. The special internal configuration of the reactor, designed and developed by Flakt through extensive laboratory and pilot testing, provides for optimum mixing of the gas and reagent. Thorough mixing and extended gas to absorbent contact time subsequently ensure increased chemical reaction and excellent absorbent utilization. Furthermore, the thorough mixing and the extensive contact time results in an agglomeration of submicron particles, such as condensed heavy metals and dioxins, to a size range that can be efficiently captured in the fabric filter. The gas, now carrying dry reaction products, flyash and agglomerates, exits the reactor and is directed to the fabric filter for collection of the particulates.

A silo is provided for storage of the fresh absorbent $Ca(OH)_2$ required for the process. The silo is designed for an effective volume of 7 to 14 days consumption. A dry material metering feeder discharges absorbent into pneumatic conveying lines equipped with blowers. The material is transported to the reactors where it is injected into the gas stream.

The dust collected in the hoppers under each reactor and fabric filter compartment is continuously discharged into a common screw conveyor system to waste or recycle.

Unreacted absorbent can still be present in the dust collected in the fabric filters. To obtain maximum absorbent utilization, a major portion of the dust would be directed to a recirculation silo from which it can be reinjected in the reactors. The recirculated dust, like the lime absorbent, is fed through ejectors to a pneumatic conveying line. This line transports the material to the reactor where it is injected into the gas stream. The system at Springfield has achieved excellent efficiency and lime utilization without the use of recycle.

The flue gas cleaning system requires about 4 man hours per shift to operate and maintain. This time is largely spent checking and correcting problems in the material handling systems. The APC system has achieved over 96% availability as a result of this attention to detail.

The Continuous Emissions Monitoring System (CEMS) at the plant was provided by Dynatron. The CEMS monitors the emissions of CO, CO_2, SO_2, NO_x, and opacity. After overcoming some software bugs during start-up the system was certified and has proven reliable and accurate.

Enercon incinerators are being operated with precipitators, baghouses, wet,dry, and semi-dry scrubbers. The preferred technology from the standpoint of gas cleaning efficiency and cost is the system at Springfield.

SERRF

The Southeast Resource Recovery Facility (SERRF) is located in Long Beach, California. SERRF is owned by a joint power authority formed by the City and the Los Angeles County Sanitation District #2. The plant serves Long Beach and surrounding cities. This waste stream averages 1180 TPD, approximately 86% of the plant's rated capacity of 1380 TPD. The project was engineered and constructed by Dravo Energy Resources, Inc. and is currently operated under sub-contract by Montenay. Total project capital cost was $106mm. The plant began operation in August of 1988.

Three furnace/boiler/APC systems make up the plant, each with a nominal capacity of 460 TPD. The steam generators share a common water treatment plant, condenser, cooling tower, and turbine.

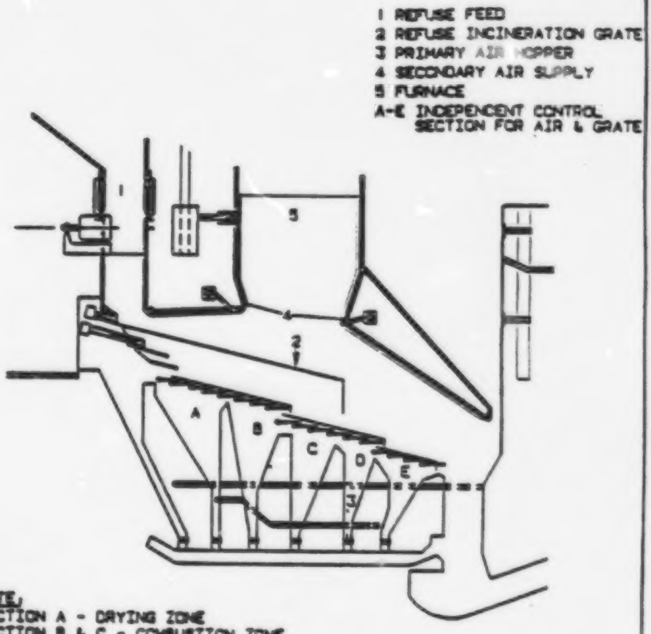
Each steam generator is designed to have an MCR of 117,170 lb/hr of steam at 650 psig and 752°F.

The plant utilizes the Steinmuller Combustion System to mass burn the MSW. The steam generators have 3 vertical radiant passes (3 pass divided by platens) followed by a pendant hung fourth pass housing 3 super heater banks, the boiler bank, and two economizer banks.

The grate is divided longitudinally into five individually operable sections labeled sequentially A through E from front to back, each of which is composed of a grate module with separate hydraulic drive. The refuse combustion grates are divided into three process stages: drying, main, and residual burnout. Each section has an individual air supply zone which can be manually balanced. Each grate section has individual dwell time control. These features allow adjustment of fuel bed agitation and air supply in accordance with the combustion requirements.

These grate zones are shown in Figure 3. After the first and third grate sections, there are agitation steps for turning over and redistributing the refuse on the grate of the following section.

The grate blocks are inclined at 12.5 degrees to the horizontal, producing, together with the hydraulically activated stoking action, a controlled forward movement of the fuel bed during combustion. Each grate module has alternate fixed and movable rows of grate bars.


1 REFUSE FEED
2 REFUSE INCINERATION GRATE
3 PRIMARY AIR HOPPER
4 SECONDARY AIR SUPPLY
5 FURNACE
A-E INDEPENDENT CONTROL SECTION FOR AIR & GRATE
1
5
4
2
A
B
C
D
E
3

NOTE,
SECTION A - DRYING ZONE
SECTION B & C - COMBUSTION ZONE
SECTION D & E - BURN OUT ZONE

L & C STEINMULLER	
MUNICIPAL SOLID WASTE INCINERATOR	
DRAVO ENERGY RESOURCES, INC.	SCALE:
JOB NO. 3798	Figure 3

The rate of fuel supply to the grates is governed by the variable idle time at the end of the stroke (rest position) and the stroke length. These are automatically controlled by the steam flow for each boiler.

The combustion air supply is divided into a primary air system, secondary air system, and a flue gas recirculation (FGR) system. Controls proportion the primary air, secondary air, and FGR flows to ensure optimum combustion and minimum NO_x generation under various operating conditions. The total airflow is controlled by inlet dampers for each fan, and the airflow apportionment to the individual zones is controlled by manual dampers and orifices in the branch ducts. The total airflow is modulated with steam flow and trimmed by the flue gas oxygen exiting the boiler to maintain the required excess air ratio.

The primary air system (also called underfire combustion air) takes up to 85% of the total combustion air at 80% excess air from the top of the refuse storage bunker. The primary air is preheated in a steam-to-air preheater prior to entering the plenums located under the grate.

The secondary air system (also called overfire air) provides up to 45% of the total combustion air required when operating at 80% excess air. The secondary air is introduced into the upper combustion zone at the throat area of the boiler through air nozzles. These nozzles are located on the front and back side of the throat. The pressure and nozzle diameter are sized to achieve at least 75% air penetration into the throat area for efficient burning. Two nozzle groups are used. For low secondary airflows every other nozzle is turned off. This ensures that the nozzle air velocity will not decrease when the air ratio is decreased. Thus penetration will be maintained regardless of the ratio of the primary air to the total combustion air.

A flue gas recirculation system is provided in which flue gases are taken from the top of the boiler economizer outlet and mixed with the primary air under zones A, B, and C. The flue gas recirculation fan normally operates at a 10% recirculation rate. The primary and secondary air flows are adjusted to maintain a constant temperature window in the furnace for the selective non-catalytic reduction system. The flue gas recirculation flow is set based on primary air flow.

FGR helps minimize NO_x formation through a number of mechanisms. By reducing the oxygen present during the initial stages of combustion, thermal NO_x formation is retarded by constraining the achievable flame temperature. The almost 15% moisture in the FGR flow is expected to enhance the formation of natural NO_x reducing agents by driving the water gas shift which produces CO. Also, by blending hot flue gas with the combustion air, the requisite pre-heat can be achieved at lower available oxygen levels than would be achieved by use of the steam coil air heater alone. To date, FGR has been seen to reduce NO_x emissions by 10-20%, but there is some evidence that CO emissions are increased.

The Exxon Thermal DeNO_x System is designed to supplement the FGR system and further reduce nitrogen oxide (NO_x) emissions from the boilers. The system involves injection of ammonia (NH_3) carried by compressed air in the "appropriate" temperature zones (1650 - 1800°F) of each boiler. Reaction of NH_3 with NO_x results in the formation of nitrogen and water in these zones. Ammonia injection manifolds are located in the side walls of each boiler. Ammonia injection rate is adjusted by feedback controls that use NO_x emissions, boiler steam flow, and NH_3 break-through or slip.

When the full plant (three boilers) steam generation is directed to the Delaval turbine it is capable of 43.5 MW. Internal plant loads are 5.5 MW resulting in an average net power sales of about 31 MW. The plant has 40 operators, 12 maintenance men, and 5 administrative staff.

The flue gas cleaning systems (as illustrated in Figure 4) for each boiler has a spray dryer and reverse air cleaned baghouse. The inlet and outlet manifolds of the baghouse are extensively cross connected so that any of the three can be used with any boiler and ID fan. The systems share a common lime storage, pumping, and slaking system. Pebble quicklime is unloaded from trucks and stored in a silo. A pneumatic conveying system transfers the lime from the long term silo to the slaker feed bin. Lime is metered out of the bin using a weigh feeder into a 2000 lb/hr Wallace and Tiernan slaker. The lime slurry produced is degritted by a 40 mesh vibrating screen and stored in an 8 hour capacity tank at 20% solids. This concentrated lime slurry is circulated around a continuous loop up to the reactor penthouses and back to the tank. A portion of this lime flow is tapped off at each reactor and diluted to approximately 5% solids before it is fed to the atomizer. The water used for dilution is cold lime softened cooling tower blowdown water. This dilute slurry is atomized into the gas stream by a single high speed rotary atomizer in each reactor. The spray evaporates and cools the gas while the lime particles neutralize the acid gases. The resulting gas is cooled from 450°F to 285°F while the slurry is evaporated to dryness. The reactor is sized for a 15 second retention time to allow for this complete drying to occur. The particulate is filtered from the gas stream in the baghouse. Screw and drag chain conveyors transfer the ash from the baghouse and reactor hoppers to the ash disposal silo. The APC residue is conditioned with water into Dravo Wellman DM-3 mixer conditioners prior to being combined with bottom ash and being conveyed to trucks.

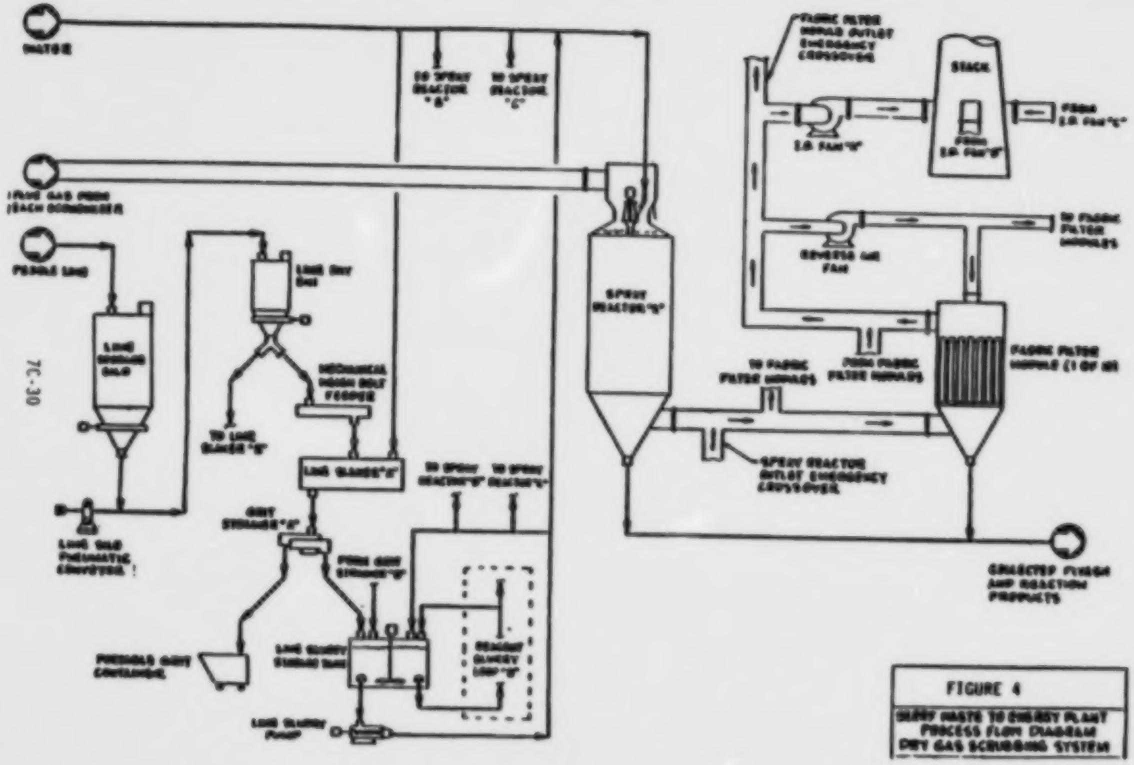

WATER
TO SPRAY REACTOR "B"
TO SPRAY REACTOR "C"
FLUE GAS FROM (EACH COMBUSTOR)
PEBBLE LIME
LIME STORAGE SILO
LIME SILO PNEUMATIC CONVEYOR
LIME DAY BIN
MECHANICAL WEIGH BELT FEEDER
TO LIME SLAKER "B"
LIME SLAKER "A"
GRIT STRAINER "A"
FROM GRIT STRAINER "B"
PORTABLE GRIT CONTAINER
LIME SLURRY STORAGE TANK
REAGENT SLURRY LOOP "B"
LIME SLURRY PUMP
TO SPRAY REACTOR "B"
TO SPRAY REACTOR "C"
SPRAY REACTOR "A"
FABRIC FILTER MODULE OUTLET EMERGENCY CROSSOVER
STACK
I.D. FAN "A"
FROM I.D. FAN "B"
FROM I.D. FAN "C"
REVERSE AIR FAN
TO FABRIC FILTER MODULES
FROM FABRIC FILTER MODULES
TO FABRIC FILTER MODULES
FABRIC FILTER MODULE (1 OF 10)
SPRAY REACTOR OUTLET EMERGENCY CROSSOVER
COLLECTED FLYASH AND REACTION PRODUCTS

FIGURE 4

SLURRY WASTE TO ENERGY PLANT
PROCESS FLOW DIAGRAM
DRY GAS SCRUBBING SYSTEM

During the first few months of operation, the flue gas cleaning system required one operator and three maintenance personnel per shift. The majority of their time was spent in discharging ash from the baghouse hoppers. Recent addition of hopper vibrators, as well as heat tracing and insulation on the screw conveyors, has nearly eliminated the problem. The atomizer temperature transmitters and the lime slaker have also required repairs and maintenance. Uneven deposits of lime grit within the atomizer have caused vibration trips. The source of these deposits is being addressed and corrected. Despite these start-up problems, the overall systems availability has been over 95% for the first 6 months. As operations become more routine, Dravo expects the O&M labor to be reduced to 1 operator and 1 maintenance man per shift.

The continuous emissions monitoring system at the plant was provided by KVB. It monitors HCl, SO_2, NO_x, CO, CO_2, O_2, and opacity at the stack. All parameters excepting opacity are also monitored at each boiler outlet for process control along with ammonia. The stack CEMS system has been certified.

Montenay also operates the Vancouver incinerator, which is equipped with dual fluid nozzle atomizers and a pulse jet cleaned baghouse. Montenay says that they prefer the Vancouver system because of the greater redundancy in reagent feed and simplicity provided by multiple dual fluid nozzles compared with a single rotary atomizer. They also feel that the particulate emissions from a pulse cleaned filter are lower than those from a reverse air type, although test results do not support this impression.

BRISTOL AND STANISLAUS RESOURCE RECOVERY FACILITIES

The Bristol Resource Recovery Facility will provide solid waste disposal services to multiple independent communities. Located on a 19-acre site adjacent to Bristol's landfill. The facility will receive and process 650 tons of waste per day (TPD) from the Connecticut communities of: Bristol, Berlin, Burlington, New Britain, Plainville, Plymouth, Southington, Washington, Wolcott, Prospect and Warren. The eleven towns provide a minimum total delivery of 153,300 tons per year to the facility.

Approximately 13 MW of electricity, generated by the waste- burning process, is sold to Connecticut Light & Power Company for distribution over its system. The facility serves approximately 270,000 people who reside within the area.

Ogden Martin owns and operates the plant. Operation began in December of 1987. Total plant cost was approximately $59mm.

The Stanislaus County Resource Recovery Facility is an 800 TPD facility located adjacent to the Fink Road landfill, which is approximately 20 miles southwest of Modesto, California.

The participating communities are Stanislaus County and the city of Modesto.

The facility generates 15 MW of electricity to be sold to Pacific Gas & Electric Company. It serves approximately 413,000 residents from the California communities of: Modesto, Oakdale, Turlock, Ceres, Hughson, Newman, Patterson, Riverbank and Waterford.

Operation began in August, 1988, and Ogden Martin Systems will own and operate the facility. Total plant cost was approximately $82mm.

The design of both plants is very similar. As shown in Figure 5, the heart of the MARTIN System for waste combustion is the MARTIN Reverse Acting Stoker Grate. The grate surface is sloped downward from the feeder end towards the residue discharge end and is comprised of alternate steps of fixed and moving grate bars. The moving grate steps perform slow reciprocating strokes against the grate slope. This ensures that the burning refuse layer is continually rotated and mingled to form an even depth of bed, and red hot mass is pushed back to the front end of the grate. Therefore, an intense fire builds up immediately at the front end of the grate, with all combustion phases (such as drying, ignition and combustion itself) taking place simultaneously and passing into one another.

In longitudinal direction, the Reverse-Acting Stoker Grate is subdivided into several zones which are supplied with primary combustion air through specially designed orifice plate openings. The primary air rate is regulated by the automatic combustion control system via sliding dampers in accordance with the combustion process.

The grate bars are made from wear-resistant and heat-proof cast steel with high chromium content. The primary air supplied to the grate bars flows through gaps, two millimeters wide between heads of adjacent grate bars, into the burning refuse layer. These air gaps form a high air resistance thus causing uniform distribution of the combustion air over the surface of each grate zone.

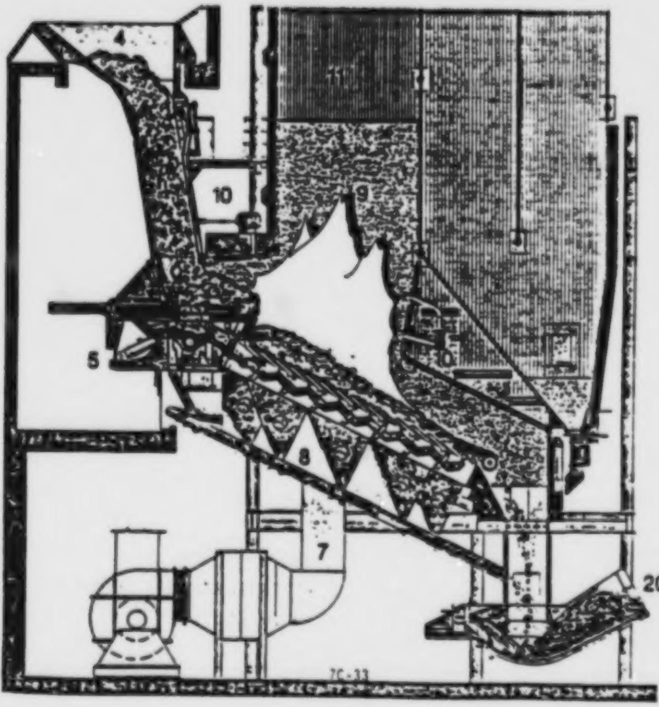
Secondary air injected at high pressure at the front and rear walls of the combustion chamber captures the approximately 1,830°F hot combustion gases above the burning refuse layer providing intense mixing, turbulence and burn-out of these gases.

Burned-out combustion residues are transferred by a slowly rotating roller at the grate discharge end into the MARTIN Residue Discharger where they are quenched and discharged.

Steam is generated at 865 psig and 830°F in both plants. The Stanislaus plant includes an Exxon Thermal DeNO$_x$ system for NO_x control by ammonia injection into the boiler.

FIGURE 5

4) Feed Hopper
5) Feeder
6) MARTIN® Reverse-Acting Stoker Grate
7) Forced-Draft Fan
8) Undergrate Air Zones
9) Furnace
10) Secondary Air Nozzles
11) Boiler
20) MARTIN® Residue Discharger



As shown in Figure 6, the APC system uses nozzle atomizing spray dryers and pulse cleaned baghouses.

The gas leaves the incinerator and boiler economizer and enters the top of the spray dryer reactor. At the top of the reactor, the gas is equally distributed to three inlet dispersers. The gas velocity accelerates as it passes through the dispersers. At the discharge of the disperser, where a finely atomized spray of lime slurry is introduced, the gas velocity is abruptly reduced. This causes a very high degree of turbulence which causes the droplets become intimately mixed with the gas.

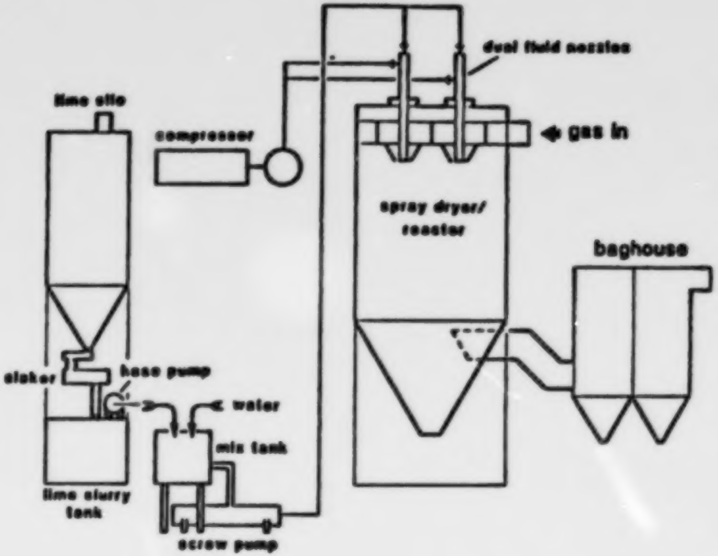
As the gas exits the disperser outlets and travels through the reactor, which at Bristol is 23 ft diameter and 42 ft high (24 x 48 at Stanislaus), the acid gas reacts with the calcium hydroxide contained in the droplets to form reaction products. At the same time, the sensible heat of the gas causes the water in the spray droplets to evaporate, thereby leaving a dry particulate residue suspended in the gas stream. Some of the dry reaction products and some of the ash fall out of the gas stream and are discharged through the reactor bottom (20-30% of the total).

The method of atomization used in the spray dryer/reactors is dual-fluid nozzle atomization. The reactor feed process slurry is mixed with compressed air in the Flakt dual-fluid nozzle. This two phase fluid is then allowed to expand isentropically through the converging nozzle. Isentropic expansion occurs because the outlet pressure is below the critical pressure of the system. The enthalphy in the compressed air (125 psig) is converted into kinetic energy or high velocity. The high velocity stream of liquid is broken into small droplets due to the drag caused by the surrounding gas.

The absorption of acid gases is enhanced because the small droplets (50 to 90 microns), which are intimately mixed into the gas stream, have a large total surface area. This large interfacial area promotes the absorption of sulphur dioxide and hydrogen chloride into the aqueous phase. The acids then react with the calcium hydroxide to form the calcium/sulphite and sulphate reaction products.

Care must be taken in the design to insure the droplet dries, neither too fast, which can hinder and curtail the absorption reactions, nor too slow, which can cause a "wet bottom" or bag blinding. The water addition rate is controlled to maintain the flue gas temperature at the spray dryer/ reactor outlet.

The fabric filter is a high ratio type with four compartments (6 at Stanislaus). The bag material is fiberglass.


lime silo
compressor
dual fluid nozzles
gas in
spray dryer/
reactor
baghouse
slaker
hose pump
water
mix tank
lime slurry
tank
screw pump

Dual Fluid Nozzle System

FIGURE 6

Quicklime from a silo is fed to a lime slaker (1 operating, 1 spare) to produce a 20% slurry. The latter is stored after degritting in an agitated tank housed in the silo structure. Metering pumps are located near the storage tank to feed the reactor feed tanks.

The flow of concentrated lime slurry to the individual reactor feed tanks is controlled by the variation from set point of the SO_2 concentration in the gas stream from the fabric filter. The reactor feed tanks are equipped with agitators to mix the lime slurry with water for dilution and variable speed positive displacement pumps to supply the reactor nozzles with diluted slurry. The reactor gas outlet temperature controls the pump speed of rotation and liquid flow.

The dust collected in the pyramidal hoppers under each fabric filter compartment is continuously discharged into a common screw conveyor.

COMPARISONS

As described previously, the systems are of widely varying size and design. Table 1 summarizes the major characteristics of the reactors and baghouses for the various systems.

The dry reactor at Springfield is very small with a short retention time. Its purpose is only to mix the gas with the lime and allow the absorption reactions to begin. The need for a long retention time for drying does not apply to the design of this type of system. The other systems have spray dryer reactors that are sized based on the requirement to produce a dry powder from a slurry spray. The nozzle type reactors are taller than their diameter while the rotary type reactor is larger in diameter than its height. These proportions are a direct result of the differing gas flow and spray patterns produced in each type of reactor. All four systems operate at similar temperatures when reacting the lime and flue gas. The spray dryers use slaked quicklime as a reagent compared to commercially dry hydrated lime for the dry absorption system.

There is a large spread in the baghouse air-to-cloth ratios used in the systems. Not surprisingly, the air-to-cloth ratio correlates well with the pressure drop and required cleaning frequency. Springfield and SERRF particularly operate with as much as 8 to 10 hours between cleanings. All the systems use woven fiberglass bags.

TABLE 1

EQUIPMENT DESIGN AND
TYPICAL OPERATING CONDITIONS

	Springfield	SERRF	Bristol	Stanislaus
Reactor				
Gas flow ACFM	24,000	118,000	85,000	94,000
Inlet temp., °F	280	500	515	520
Outlet temp., °F	280	285	285	285
Dia. x Height, Ft.	5.5x17	33x22	23x42	24x48
Retention time,Sec.	1	15	15	15
Atomization	none	rotary	nozzle	nozzle
Type lime	hydrated	slaked	slaked	slaked
Baghouse				
# compartments	4	10	4	6
#bags	504	1680	1064	1596
bag size dia x len.	6"x14'	8"x22'	6"x14'	6"x12'
bag material	--------teflon	coated	fiberglass--------	
Gross A/C	2.2	1.3	3.2	2.8
Net A/C	2.9	1.5	4.2	3.2
Cleaning	pulse	rev. air	pulse	pulse
Operating Data* (per boiler)				
Lime, lb/hr	83	165	160	208
Pressure drop, "wg	6	9	13	8
Power, kW	25	160	85	89
Labor, mh/day	4	16	6	6

* Actual operating conditions vary substantially based on boiler load and fuel composition

The stoichiometric ratio is comparable in all cases, although, the actual lime consumption varies somewhat because of differences in the acid gas content in the flue gas. Springfield uses hydrated lime with a higher molecular weight and thus a higher lb/hr feed rate for the same amount of calcium.

The pressure drop variation reflects both air-to-cloth ratio and the fact that dry reactors require less pressure drop to operate than do spray dryers.

Power consumption on a per ton basis is comparable with the exception of SERRF. The use of a rotary atomizer and reverse air fan increase the power consumption of that system compared to the others.

The high O&M labor rate at SERRF reflects start-up and design conditions and will likely drop to a level similar to the others over time.

The emissions test results for the various systems are shown in Table 2. Reasonably comparable results are achieved for acid gases, particulates, metals, and dioxins, regardless of the type of APC system used. There are dramatic differences in the NO_x and CO emissions of the boilers depending on their design or the type of NO_x control used.

The costs of the APC Systems on the various plants is indicated in Table 3. Costs were normalized to a basis of capital cost per ton per day of capacity and operating costs per ton of garbage burned. Springfield shows the highest capital cost per ton because of its small size. In that size range, a spray dryer would have been even more expensive. SERRF is a relatively expensive plant because of the large reverse air baghouse used. Bristol and Stanislaus are a typical size and design used at many other plants. The operating costs are strongly influenced by debt service on the capital cost of the plant, thus Springfield and SERRF are expensive from this aspect also.

Considering the variance in the size and design of the plants, it is remarkable that the capital and operating costs for these systems are so similar. Any of these APC technologies could be selected depending on special requirements for a specific installation. They have all proven effective and economic techniques for pollution control on incinerators.

TABLE 2

EMISSIONS TEST RESULTS

		Springfield	SERRF	Bristol	Stanislaus
Gases		@ 12% CO_2	@ 3% O_2	@ 12% CO_2	@ 12% CO_2
HCl	inlet ppmd	503	*550	383-539	**440
	outlet ppmd	31	31	4-15	** 50
	efficiency %	94	* 94	96-99	** 89
HF	inlet ppmd	.9	n/a	n/a	** 22
	outlet ppmd	.08	.07	<.1	** 3
	efficiency %	92	n/a	n/a	** 86
SO_2	inlet ppmd	129	179	*100-150	**130
	outlet ppmd	22	9	4-48	** 30
	efficiency %	83	95	*80	** 77
NO_x	outlet ppmd	143	**150	280	**200
CO	outlet ppmd	21	n/a	19	n/a
Particulate					
inlet gr/scfd @ 12% CO_2		.9	1-1.9	*.3	**2.5
outlet gr/scfd @ 12% CO_2		.0016	.005-.008	.0021-.0113	**.011
Metals					
outlet mg/NM_3		@ 12% CO_2	@ 3% O_2	@ 12% CO_2	
Mercury		.30	.23	.11	n/a
Nickel		.024	.0034	<.005	n/a
Lead		.021	<.02	.03	n/a
Tin		.021	n/a	n/a	n/a
Chromium		.01	.0039	<.002	n/a
Cadmium		.001	.023	n/a	n/a
Arsenic		<.0007	<.0039	.003	n/a
Beryllium		<.0001	<.0079	8E-6	n/a
Antimony		<.005	<.0039	n/a	n/a
Selenium		<.0007	<.0039	n/a	n/a
Vanadium		<.001	n/a	n/a	n/a
Dioxin outlet ng/NM_3 2, 3, 7, 8 TCDD equiv. EPA Method		.15	n/a	.09	n/a

* Typical operation
** Design value
n/a Data not available

TABLE 3

ECONOMIC COMPARISON

	Springfield	SERRF	Bristol	Stanislaus
Capital Cost				
Plant size, TPD	3x120	460	2x325	2x400
APC cost, mm $	3.0	10	4.0	4.7
Unit cost, K$/TPD	8.3	7.6	6.1	5.9
Operating costs per ton of waste				
1)Debt service	$2.80	$2.50	$2.00	$2.00
2)Lime	$0.70	$0.30	$0.40	$0.40
3)Pressure drop	$0.40	$0.70	$0.90	$0.60
4)Power	$0.50	$0.90	$0.60	$0.50
5)Labor	$0.70	$0.70	$0.40	$0.30
Total	$5.10	$5.10	$4.30	$3.80

0) All calculations based on operation 300 days per year at design load
1) 10% cost of money
2) Hydrated lime @ $85/ton, pebble lime @ $65/ton
3) ID fan power calculated based on 70% fan efficiency
4) $.10/kW
5) $20/hr

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UTILIZATION OF FLUE-GAS CLEANING TECHNOLOGIES FOR WASTE INCINERATORS IN EUROPE

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ABSTRACT

This paper presents an overview of a substantial ongoing air pollution control program activity in West Germany and other western European countries as it relates to acid gas, trace metals and products of incomplete combustion emitted from waste incineration. Details are given of emission regulations, control means used, and technical advancements accomplished and foreseen.

European emphasis on utilization of wet scrubbing methods to maximise pollutant collection efficiencies is reviewed and examined for the purpose of outlining and quantifying gains achieved. Related European use of flue gas sub-cooling means is also assessed to describe benefits in stack gas cleaning performance.

Techniques are reviewed for minimizing environmental impact in the management of incinerator residue as well as solid and liquid wastes from the flue gas cleaning.

Advanced commercial practices are highlighted that appear to offer the greatest potential for technological contribution to the ongoing, widespread expansion in U.S.A. of municipal solid waste and hazardous waste incineration facilities.

INTRODUCTION

Significant design and operating experience in stack gas emission control and management of collected wastes at municipal waste incinerator facilities has been gained in Europe over the past 20 years. Approximately 180 such incineration systems, each with a capacity in the range of 200 to 960 metric tons per day of municipal refuse, are in operation in Europe, principally in West Germany, France, The United Kingdom, Switzerland, Italy, Sweden, Belgium, The Netherlands, and Denmark. West Germany is the leading European country in the amount of municipal waste incinerated, the use of heat recovery facilities, and the number and efficiency of flue gas cleaning facilities. Approximately 8.5 million metric tons of waste per year are presently incinerated in 47 waste incineration plants in West Germany and 21 million inhabitants (corresponding to 34% of the total population) are served by these plants. (The total amount of annual municipal waste is approximately 32 million tons per year). Over a recent 15 year period there has been a 180% increase in incineration capacity (to 8.5 million metric tons/year); a 90% increase in the number of incineration plants (to 47); a 150% increase in the number of inhabitants served by incineration (to 21 million); a 15% increase in the volume of incinerated waste per inhabitant served, to 380 kg per year per inhabitant served (2.3 lbs/da/person); and a 50% increase in the average capacity of existing plants to 175,000 metric tons per year (530 short tons per calendar day). It is anticipated that by the end of 1995 an additional 10 MWC plants will have come into operation.

REGULATIONS

WEST GERMAN REGULATIONS

The 1974 West German Waste Act rev. August 27, 1986, requires that priority be given to minimization of waste generation through use of minimum-waste-producing technologies and waste utilization, and that waste disposal be ranked as the lowest priority solution (1). Article 7 of the Waste Act requires that the installation and operation of MWC (municipal waste combustion) plants be subject to an official general planning procedure covering the entire path of the waste management process. MWC-plants with a capacity equal to or greater than 0.75 metric tons per hour are subject to a formal licensing with public participation; smaller plants are licensed by a more simple non-public procedure.

The requirements of the 1974 Federal Environmental Quality Control Act (Bundes-Imissionsschutzgesetz-BImSchG) revised April 24, 1986, which apply directly to incinerators call for control of emissions by the best available technology. Environmentally suitable management of incinerator residues is also emphasized.

The best available technology for emission control is defined in the (2) "Technical Instruction for Air Pollution Control" (TI Air), a general administrative regulation. The emission standards of the TI Air for MWC plants are shown in Table I, wherein Nm^3 represents a standard cubic meter at ref.conds.

Existing plants must also meet these emission standards. The retrofitting of emission reduction technology has to be performed within certain time periods of 3, 5 or 8 years, (from March 1, 1986), depending on the environmental risk potential of the emission source. For instance, MWC plants without any HCl- or HF- emission reduction device have to be retrofitted within 3 years. Existing-source SO2 and dust emissions must meet the standards within 5 or 8 years, but in some instances the time schedule is 3 years, e.g. if dust emission is more than 3 times the limit value.

EUROPE WIDE REGULATIONS

Typical Stack Emission Limits by Recent Enactment

Table I presents emission limits applicable to existing and new municipal solid waste incinerators in western European countries. This reflects a strong and growing movement toward limiting emissions of products of incomplete combustion (PICs) such as dioxins and furans.

1987 Action in Sweden on PICs

Introduction -- Polychlorinated dibenzo-para-dioxins (PCDD) and polychlorinated dibenzofurans (PCDF) are two groups of chemicals collectively called dioxins, several individual forms of which are extremely toxic. They are chemically stable and break down only slowly in the natural environment. Although there has been considerable research into their effects on health and the environment, the results provide a very uncertain basis on which to estimate risk. Accordingly, and within the policy of the National Swedish Environmental Protection Board (3), however, it is felt that enough is known to require that dioxins be regarded as a definite hazard; a major program is underway in Sweden to limit the release of these substances to the environment. Fuel combustion processes - especially when organic chlorine compounds are present - are seen to always constitute a potential source. The less efficient the combustion, the greater will be the emissions of dioxins. Waste incineration is also a major source. The emissions from all such existing plants are to be greatly reduced by 1991. Dioxins can be transported over great distances, indicating a regional, and possibly global, distribution. This is seen to make them an especially serious threat to the oceans, since pollutants tend to accumulate in these bodies. Dioxins are stable organic substances that are only slowly broken down in the natural environment. When emitted they can therefore disperse over a wide area and continue to act for a long time.

Terms of Measurement of Dioxins -- There are 75 different polychlorinated dibenzo-para-dioxins and 135 polychlorinated dibenzofurans. The molecules are flat, tricyclic aromatic hydrocarbons, all with similar chemical characteristics. They vary greatly, however, in toxicity and biological properties. The best known and most studied of these substances is 2,3,7,8-tetra-chloro-dibenzo-para-dioxin, or TCDD. This is probably also the most toxic of them all. TCDD equivalents are means of summing up the toxic effects of different PCDDSs and PCDFs in order to calculate the toxic effect of mixtures. The amount of each such substance is converted to the amount of TCDDs that would produce the same effect. The conversion factors are called TCDD equivalent factors (TEF), and a substance that is half as toxic as TCDD has TEF = 0.5.

Known Emission Inventory -- In terms of TCDD equivalents, the emissions of dioxins in Sweden is believed to total approximately half a kilogram (500 grams) a year. This estimate is however based on somewhat uncertain data. Such information as is now available regarding the inventory of Swedish emissions of dioxins is summarized in Table II. Since the values are based on single field measurements, these yearly estimates are also uncertain. Moreover, other potential sources are being investigated.

New Swedish Emission Limits -- The Swedish government has in 1987 sanctioned a major program proposed by its Environmental Protection Board and National Energy Administration for reducing dioxin emissions from incinerators, mandating that every existing plant in the country shall be made capable of fulfilling the new requirements by the end of 1991. Extensive capital investments are now being made in Sweden to achieve these objectives, summarized as follows:

- At the time of inspection, emissions from existing plants should not exceed the range of 0.5-2.0 ng/Nm^3 (dry gas basis with 10% CO2).
- For new (future) plants the corresponding level should be 0.1 ng/Nm^3.
- The dioxin values are in TCDD equivalents and measured during normal operation in accordance with the recommendation of an official Scandinavian dioxin group.
- The stated values are to be taken as guides during an initial, trial period. Definite limit values are to be set, in each case, at the end of the trial period.

STACK GAS CLEANING METHODS

All MWC plants in West Germany have a dust separator installed - in most instances electrostatic precipitators, in others fabric filters or a combination of these two - which typically reduces dust emissions below 100 mg/Nm^3 or, at new plants, below 30 mg/Nm^3. About 2/3 of the incineration capacity is equipped with additional flue gas treatment technology for the reduction of emissions of HCl, HF and heavy metals. Up to recently there has been an almost equal balance between wet scrubbing vs. spray drying and dry absorption processes. The choice between wet vs. dry/semi-dry scrubbing has been greatly influenced by both state/community criteria dealing with disposal of residues and wastewater management as well as system cost (4). In the future a trend to wet scrubbing is expected because of greater control of SO2, heavy metals, and other toxic substances. Moreover, spray drying and dry processes have the disadvantage of greater residue production. But on the other hand, wet scrubbing presents wastewater management problems.

Liquid effluent treatment by heavy metal precipitation and/or evaporization are said to be promising solutions. Currently constructed MWC-plants in West Germany utilize special combustion technology (minimizing the formation of toxic hydrocarbons and dioxins), with high efficiency dust separators, and with two-step wet scrubbers (first step: heavy metals, HCl, and HF;collection: second step: SO2 collection).

Available European-wide data indicate that lime spray dryer/ESP and spray dryer/fabric filter systems can attain 70-90% acid gas removal and 97% or more control of dioxins and furans while limiting mercury emissions to about 0.01-0.07 mg/Nm³ (dry). In comparison, some wet scrubber system designs can attain 90+% acid gas removal, have potential for removal of NOx and can achieve control of dioxins and furans comparable with semi-dry scrubbers while possibly providing consistently lower mercury emissions (5).

WET SCRUBBERS

Spray tower and venturi wet scrubbers have been commonly used downstream of ESPs to remove HCl, HF, and SO2. HCl and HF are frequently removed in the first scrubber stage (or prescrubber), and SO2 is collected in a separate second scrubber stage.

Sub Cooling -- In some installations, particularly in Sweden, a heat exchanger is included downstream of the wet scrubber to condense water vapor from the water-saturated, cleaned flue gas. This heat recovery/removal step, which is termed subcooling, reduces the visible steam-plume discharge and improves removal of submicrometer (submicron) particles and metallic vapors. This sequencing differs from that utilized in many hazardous waste incinerator applications in U.S.A. (6), wherein the subcooling/condensation by recirculated cooling water takes place after a raw-gas water-quenching step, but before wet particulate removal facilities.

Liquid Effluent Management -- Some installations have a wet scrubber system downstream of a spray dryer that receives the wet scrubber effluent and converts it to a solid waste, thus avoiding liquid effluent discharge. The spray dryer in this configuration is designed to dry the wet scrubber effluent without significant collection of acid gases.

Reheating of Exhaust Gas -- As in the case of coal-fired boiler applications, reheating of water-saturated flue gas is frequently needed in conjunction with use of an existing incinerator stack by retrofit scrubber facilities. Extensive use is being made in the European incinerator cleanup of reheat exchanger designs of non-metallic construction to resist high corrosion potential, which is particularly great when raw-gas heat recovery is employed (7). Where low-cost hot water or steam can be made available, a PFA-Teflon lined tube bundle heat exchanger has been used for the flue-gas reheating, with the gas flowing outside the tubes and the heating medium inside the tubes. Teflon lining is necessary because the clean gases from the scrubber carry over a significant amount of scrubbing liquor mist, which would corrode unprotected (metallic) heat exchange surfaces. In some applications a gas to gas reheat exchanger is used to utilize heat scavenged from the raw gas. A single, cross-flow, tube bundle heat exchanger is being used with the raw gas flowing vertically inside the tubes and the clean gas horizontally outside the tubes. In the initial commercial development of equipment for this service glass tubes were used as heat-exchanger surface material, but after several years of operation it was seen that the HF-content together with the SO_3 is excessively/corrosive to the glass at surface temperatures up to 140°C (284°F). To overcome this problem replacement tube materials used include PTFE-Teflon.

NOx Removal -- Scrubber systems are in many instances designed to remove NOx as described in the following section on Key Pollutants in Stack Gas.

SPRAY DRYER SCRUBBERS

In the common semi-dry system operation, the raw flue gas is contacted in a spray dryer with a cloud of fine slaked-lime slurry droplets from a rotary atomizer or dual-fluid nozzles.

Applicability -- Employed in conjunction with a downstream fabric filter or ESP, HCl, SO_2, and fine particulates can be removed efficiently and effectively. Approximately 30% of all municipal solid waste incinerators in West Germany use semi-dry scrubber operation for acid gas and particulate removal (4).

Influence of HCl -- Due to the high HCl concentration in the flue gas and the hygroscopic characteristics of the dry $CaCl_2$ formed from it, acid gases (HCl and SO_2) are absorbed at high efficiency by semi-dry systems like spray dryer absorbers (or dry injection systems using lime as sorbent).

DRY LIME INJECTION

Effective dry absorption of acid gases is possible through pneumatic injection of hydrated lime solely or in combination with recirculated dust-catch into raw flue gas cooled to 110-170°C (230-335°F). A circulating fluid-bed absorption system has in some applications been used for dry absorption in incinerator service, providing significantly greater sorbent residence time (than spray dryers) upstream of an ESP. Approximately 20% of all municipal solid waste incinerators in West Germany use such totally dry scrubbing operation for acid gas and particulate removal (4).

KEY POLLUTANTS IN STACK GAS

PARTICULATE MATTER

With the traditionally used ESPs, very low particulate emission levels are achievable (5) less than 45 mg/Nm^3 (0.02 gr/dscf) at a high ratio of collector plate surface area to gas flow volume [i.e., in the range of 170 m^2-min/m^3 or min/m (52 min/ft) or greater].

Due to the fine size of incinerator particulates, normal wet venturi scrubbers are relatively ineffective for particle emission control, removing 80-95% in normal operation. Very high gas-pressure-drop is required to remove fine particulates; hence, these scrubbers are not generally used as the principal particulate control means.

Fabric filters were for many years used sparingly because of the perceived failures caused by hot gases, spark carryover, or sticky particles. However, fabric filters can control particulate to 45 mg/Nm³ (0.02 gr/dscf) or less without operational upsets due to varying fuel and ash composition, which may adversely affect ESPs. With significant use of spray dryer scrubbing systems in Europe,particulate emissions from incinerators are being controlled by highly efficient fabric filters or ESPs.

Field survey work at a number of European dry and semi-dry scrubbing installations in western Europe (8) reported that there were significant fabric-filter operating problems with caking up, blinding and burnout-failure of filter bags. Continued surveillance of incinerator fabric filter reliability is being carried out.

ACID GASES

The West German inventory of HCl emissions from waste incineration peaked in 1979 at approximately 24,000 metric tons per year and by the late 1980s is below 12,000 metric tons per year. SO2 emissions reached 16,000 tons in 1981 and is understood to now be below 12,000 tons. Gross incinerator emissions, 11% O2 basis, typically include HCl at a concentration as high as 500 ppm, SO2 up to 220 ppm and HF up to 9 ppm.

Control of acid gases (HCl, HF, and SO2) requires scrubbing or devices for gas/liquid or gas/solid contact. Water alone is a reasonably effective sorbent for very reactive acid gases such as HCl (and HF) in the first scrubber stage, but an alkali sorbent (i.e., control of liquid pH to 5 or higher) and additional liquid/gas contact time in a second stage are necessary for substantial SO2 control (5). Sodium hydroxide rather than lime/limestone is most often used in the second wet scrubbing stage to avoid calcium scaling. (The sodium sulfite reaction product is water soluble as contrasted with calcium sulfite, which is highly insoluble). Highly efficient control of acid gases may be achieved by alkali wet scrubbers operating at saturation (65°C or 150°F outlet temperature), but typically requires the management of liquid blowdown.

Use of totally dry sorbents, on the other hand, requires substantial residence time in the gas for effective acid gas control. To be effective in this mode, injection of sorbent into a duct must be complemented by a fluid-bed reactor, humidification, a fabric filter dust collector, or a combination of these.

Spray drying or semi-dry use of sorbent is more effective than dry injection, with increasing acid gas removal as the outlet gas temperature decreases toward the water saturation temperature. It appears that the existing spray dryer absorbers at incinerators operate at an outlet temperature in the range of 120-160°C (248-320°F), while dry injection systems without gas humidification operate at a higher gas temperature. Both the spray dryer absorber and dry sorbent injection into the flue gas duct are followed by an ESP or a fabric filter for particulate removal. Note that the performance of an ESP actually appears to improve when located downstream of a spray dryer, probably due to the reduced gas volume and gas particle changes resulting at the lower gas temperature. The approximate acid gas removal capabilities of many gas cleaning arrangements that have been applied or proposed is for HCl, 95+% for wet and semi-dry modes, 80-90% for dry mode systems; for SO2, to 90% and higher for wet systems and for semi-dry systems utilizing sorbent recycle, and as low as 50-70% for semi-dry systems without sorbent recycle and using an ESP for particulate collection, and 50 to 60% for dry mode systems; for HF, typically 99%, and with as low as 98% for dry mode systems.

PRODUCTS OF INCOMPLETE COMBUSTION (PICs)

By optimization of the combustion (temperature, turbulence, time and oxygen) and by keeping boiler surfaces clean the concentration of dioxins can be reduced considerably (9). Many investigations have been made showing significant formation of dioxins in oxygen-rich flue gases at approximately 325°C (617°F), with catalysis of the phenomena by some heavy metals.

Capture of dioxins and furans, as wellas other trace organic compounds by condensation and capture as particular matter is significant along with reaction and capture by alkali (5). Control is enhanced by lowering flue gas temperatures, subjecting flue gas to caustic sorbent, and removing the product in a highly efficient particulate collector. Spray drying followed by fabric filtration is very effective for such organic vapor control and superior to a spray dryer/ESP system (10). Also, lower flue gas temperatures favor increased removal. Pilot plant results indicate that many of the dioxin species are removed at an efficiency of only 50 to 90% when the fabric filter equipped spray dryer system is operated at a collector temperature as high as 160°C (320°F) but that most species are removed at greater than 99% efficiency at 120°C (248°F).

The significance of condensation in removal of dioxins and furans is emphasized in recent municipal solid waste incinerator gas cleaning tests in Oregon (11). With spray-dryer absorber outlet temperature carried as low as 127°C (262°F) in some tests total CDD and CDF in the fabric filter ash over the range of test operations were typically 10-15 ppb as compared to 2-10 ppb in other ash samples upstream of the gas cleaning system. It was reported that the long collected-particulate residence time in the baghouse (8 hour compartment cleaning cycle for bag dedusting) and low flue gas temperature probably aided solids collection on the filter fabric surfaces.

HEAVY METALS

The strategy for control of emissions of heavy metals, which in Europe has attracted at least as much attention as have PICs, is similar to that for control of organic pollutants in that effective particulate capture and low flue gas temperature are major elements. Sorbents, however, are not seen to play a major role in heavy metals capture. Toxic metals enter the collectors as solids, liquids, and vapors and, as the flue gas cools, the vapor portion converts to collectible solids and liquids. However, very volatile metals, particularly mercury, are present as vapor in measurable amounts even at the lowest outlet temperatures at which collectors are designed and operated. Note that the flue gases leaving the incinerator boiler contain mercury in two states: as a vapor and as a solid. At flue gas temperatures of about 400°F (200°C) a large portion (about 95%) of the mercury is present as gaseous mercury chloride ($HgCl_2$), which is very soluble in water and hence is efficiently removed by wet scrubbers.

Additionally, the effectiveness of heavy metals emission control by diverse gas cleaning system designs appears to be inversely related to system outlet temperature. Testing in Sweden of a pilot wet scrubber downstream of an ESP indicated that, with subcooling of saturated flue gas to approximately 60°C (140°F), the total (particulate and gaseous) mercury emissions from a municipal waste incinerator were reduced to as low as 0.01 mg/Nm^3 (dry) (12). Tests of a German municipal refuse incinerator equipped with a spray dryer followed by an ESP operating at 150°C (302°F) indicated that vapor-phase mercury emission alone was 0.05 mg/m^3 or greater, (11% O_2 basis) (14). A dry injection facility serving a municipal waste refuse incinerator in Sweden, with its downstream fabric filter operating at approximately 160°C (320°F), is reported to reduce particulate mercury to a typically non-detectable concentration and gaseous mercury to the range of 0.012-0.065 mg/Nm^3 dry (11% O_2 basis).

In systems in which wet scrubber liquid effluent, suitably neutralized, is converted to a solid waste form in an upstream spray dryer system, mercury dissolved in the liquid is re-evaporated thus regenerating the gas borne mercury (13). To avoid this mercury buildup selective ion exchange system is utilized to treat the scrubber effluent before neutralization so as to remove the mercury compound.

NOX

The West German inventory of NOx emissions from waste incineration has peaked in the late 1980s at approximately 11,500 metric tons per year. Gross incinerator emissions, 11% O_2 basis, typically include NOx (13) at a concentration that may range above 350 mg/Nm^3 (175 ppm). Although these levels are generally less than the new TA Luft limit of 500 mg/Nm^3 (250 ppm) (14),some new incinerator installations are providing for NOx removal because of site-specific factors, including imposition of more stringent local/provincial emission regulations. In some instances requirements for NOx removal are the result of European Economic Community agreement in 1987 for nation by nation reductions in overall NOx emission inventories during the late 1980s and early 1990s.

Removal by Wet Scrubbing

NOx is considered to probably be the most difficult and expensive pollutant to control, primarily due to the low reactivity and water solubility of NO, which comprises 95% or more of the total uncontrolled NOx emission. Wet scrubbing technologies applied or contemplated in Europe for NOx removal include wet absorption in either the oxidation-absorption or absorption-reduction mode.

Oxidation-Absorption -- In a wet scrubber system operating downstream of a spray dryer an oxidizer, such as sodium chlorite ($NaClO_2$), is added to oxidize flue gas NO to NO_2, which can be removed by a sodium-base wet scrubbing operation (15). For example, with the addition of $NaClO_2$ to an upstream venturi (HCl) scrubber and raising the pH to 3-4 some NO is oxidized to NO_2 in the first venturi scrubber and is thereafter removed in this scrubber, and along with SO_2 in the following venturi scrubber.

Absorption-Reduction -- Another approach used to deal with the limited reactivity of flue gas NO and its poor absorption in typical wet scrubber operations is absorption-reduction. This method of NO removal has been forecast in 1987 for municipal refuse incinerators in Bremerhaven, West Germany (16). In this system, ferrous ions (Fe^{2+}) tie up NO in the liquid phase by the formation of ethylene diaminetetraacetic acid (EDTA) complexes. These complexes, in turn, react with HSO_3^- and SO_3^{2-} ions (from SO_2 absorption), leaving N_2 and sulfate SO_4^{2-}) as reaction products. Chemical additive cost for this technology is reported to be approximately 4-11 DM (approximately $2-5 U.S.) per metric ton of waste processed) less than that for oxidation-absorption processing for NOx removal (17). Reports in 1988 indicated that due to major performance failure and operational problems of an absorption-reduction scrubbing system on a large, West German, (800 MWe), coal fired boiler, use of this wet absorption mode had been prohibited at Bremerhaven.

Dry Removal Methods

NSCR -- Non-catalytic selective reduction (Exxon Process) utilizing ammonia as a reducing agent has been scheduled for use at the Neustadt (Schleswig-Holstein) incineration plant (4).

In this injection process ammonia reacts with NO at 850-1,000°C (1560-1830°F) to yield N_2. At temperatures above this range an excessive amount of ammonia feed is oxidized to NO or N_2. The temperature window for the optimal reaction is very narrow and is not always located in the same place in the boiler. It is, therefore, necessary to install nozzles for the injection of NH_3 solution at several levels in the boiler. Measurement of a reference temperature in the gas stream assists in locating the proper height for NH_3 injection.

The amount of NH_3 injection is controlled in response to the NOx concentration in the stack. In conjunction with a facility that uses a wet scrubber the process can satisfactorily operate at the high NH_3/NOx stoichiometric ratio required at the low end of the temperature window, (while benefiting from the very low rate of ammonia loss through oxidation at this temperature level). This is possible where a system is employed to recover ammonia from the scrubber liquor.

The Exxon installation is now reported by Von Roll to have been made at the Bremerhaven incineration plant, each train of which processes 360 metric tons per day, (originally earmarked for absorption-reduction scrubbing of NOx), utilizing excess ammonia feed (ammonia slip) recovered by the wet scrubber gas cleaning system (13). With a flue-gas ammonia concentration of 100 mg/Nm^3 and 95+% capture in the wet scrubbing system, the net emission of ammonia is below 5 mg/Nm^3. The collected ammonia is stripped from the scrubbing liquor with steam and is reinjected into the boiler. Commencing operation in January, 1988, the Bremerhaven gas cleaning installation is reducing inlet NOx from the range of 300-400 mg/Nm^3 (150-200 ppm) to 70 mg/Nm^3 (35 ppm).

KRC Two-Stage DeNOx Process -- A low ammonia-slip, non-catalytic dry NOx removal system that requires no downstream wet scrubbing system has been installed by KRC Umwelttechnik in 1988 at a 330 metric ton per day furnace train of the KVA Basel municipal solid waste incineration plant in Basel, Switzerland (18) in a facility that has no scrubber. The process, which reduces NOx by 65 to 80%, injects urea solution in a boiler heat transfer section that fluctuates in temperature between 815 and 1040°C (1,500 and 1,900°F), liberating ammonia that reduces NOx. Liquid methanol is injected in the rear cavity of the boiler to significantly reduce the substantial ammonia slip that would otherwise exit the stack. The process is capable of limiting the ammonia slip to 5 ppm or less.

INCINERATION WASTE MANAGEMENT

By the late 1980s production of bottom ash from waste incineration in West Germany has reached 2.5 million metric tons. Solid waste from flue gas cleaning now totals approximately 50,000 metric tons per year of wet scrubber sludge, 250,000 metric tons per year from dry and semi-dry sorption systems and 100,000 metric tons of dry fly ash catch. Annual fly ash production reached a peak of 175,000 metric tons in 1986 prior to extensive replacement-application of dry and semi-dry scrubbers.

INCINERATOR RESIDUE

Incineration produces ash and slag containing dioxins. Based on tests in Sweden (3), however, it has been shown to be very difficult to leach out any dioxin from such residues using water. With careful land disposal and monitoring ash and slag should therefore not constitute any serious source of discharge to the environment.

FLUE GAS CLEANING WASTES

Solids

For the treatment of collected fly ash to minimize hazardous waste production two new procedures have been investigated in West Germany (1). One procedure uses a recirculation of the dust into the firebox after having leached out the heavy metals content. The other more promising procedure consists of thermal treatment of the dust under oxygen deficient conditions at temperatures of approximately 300°C (572°F). 96% dioxin destruction is reported after 1 minute residence time, 100% after 5 minutes residence time.

Liquids (13)

Two basic wastewater processing techniques are used in Europe in managing wet scrubber effluent.

Spray Drying -- After leaving the boiler, the flue gases at a temperature of 200 to 280°C (400 to 530°F) flow through a spray dryer. Wet scrubber blowdown, which has been neutralized with lime, is injected into the hot flue gases in the spray dryer. The necessary heat for the evaporation of the spray dryer feed is supplied by sensible heat of hot flue gases. The flue gases are, thereby, cooled to a temperature of 160°C (320°F). The behavior of the dry residues and the potential for corrosion (acid condensation) require that the operating temperature of the spray dryer be maintained above approximately 160°C (320°F) (25). Due to the characteristics of the waste being burned and the resulting particulate, some of the acid gases, such as HCl, HF, and SO_2, adsorb onto dust particles and droplets in the spray dryer. The dry powder removed in the downstream particulate separator is discharged via an airlock valve. It consists of salt, fly ash, and heavy metals in hydroxide form.

Treatment and Outfall Discharge -- The neutralizer scrubber blowdown from the flue gas cleaning system flows to a wastewater treatment system. Through flocculation and precipitation, sludge containing heavy metals such as hydroxides, fly ash, and insoluble salts, is separated in a settling pond. Subsequently, this sludge is dewatered and disposed of in a secure landfill. The saline water overflow is discharged to the sewer. The amount of residues which have to be disposed of is typically much less in the case of the wet method than for the spray dryer technique above.

NEW TECHNOLOGY

Development of the new Multi-Recycle (MR) Process is reported in Austria (19). Flue gas cleaning is carried out in a two stage wet scrubbing system that utilizes alkaline constituents of bottom and fly ash to efficiently capture and neutralize flue-gas acid gases. The scrubbing liquor thus leaches essentially all soluble constituents from the bottom and fly ash including both alkaline compounds and heavy metals. Final treatment of the ash to accomplish this is carried out using a horizontal vacuum belt filter with the ash being washed, counterflow, and with increasingly acidic scrubbing liquor, pH decreasing to approximately 1. Prior to discharge of the treated ash at the highest landfill quality,suitable for use as construction material, it is washed with fresh water, the filtrate being recycled to the scrubbing facility. Gypsum is precipitated as the main product of the scrubbing operation and is removed from the system together with the bottom and fly ash product.

After the wet scrubbing step the flue gas passes to a wet electrostatic precipitator for final particulate removal. When needed an SCR facility follows for removal of NOx.

Organic toxic residues, dioxins and furans are essentially all adsorbed on the surface of fly ash particles in the cooler sections of the flue gas path (316°C, i.e. 600°F, last boiler section, and ESP). This ash is collected separately from bottom and high temperature ash. Subsequently this ash, which amounts to less than 1/10 of all the solids collected undergoes a separate thermal treatment process in a rotary kiln. All solid residues from the combustion process are claimed to exhibit excellent quality in regard to leachability and organic toxic substances. The material can be utilized as building material.

The heavy metals leached out of the ash by the acidic scrubbing liquor are removed from the system by treating a slip stream of the scrubbing liquor in a wastewater treatment facility. By adding neutralization agents the heavy metals are precipitated as hydroxides that are subsequently removed from the liquor in a flocculation and dewatering stage. The filter cake thus removed from the system is very rich in heavy metals but devoid of gypsum present in sludge from most conventional systems. Although the composition of this solid waste qualifies it as a hazardous waste, it is so concentrated in heavy metals such as zinc, lead and cadmium that it constitutes an attractive raw material for reclaiming of selected metals by appropriate processing. The liquid effluent from the process is an inorganic liquor the principal dissolved solid constituent of which is highly water soluble calcium chloride. Where a suitable receiving stream is not available for outfall discharge an evaporation process may be used to render it as a salt cake for disposal or use in road deicing.

TECHNICAL ADVANCEMENTS FORESEEN

U.S. APPLICATION OF ADVANCED PRACTICES

Municipal Solid Waste

The European incineration market, particularly in West Germany, Austria and Sweden,has been pursuing the best available control technology for waste incineration flue gas cleaning for the last 5-10 years. Activities in these countries may provide important guidance in ongoing application of advanced practices in U.S.A. Two different processes each have gained broad commercial application: (a) two-stage wet scrubbing for acid gas removal, which may use the sensible heat in the flue gas for upstream spray drying of the wet scrubber liquid waste stream; and (b) lime dry absorption in humidified furnace flue gas or lime spray-drying absorption for acid gas control, followed in either case by particulate collection using an ESP or a fabric filter.

While wet scrubbers may achieve slightly lower emissions of acid gases (HCl, HF, and SO_2) than spray dryer absorbers, this appears less important than other emerging emission concerns, such as heavy metals and organics (5).

Spray dryer absorbers followed by ESP's or fabric filters have demonstrated satisfactory results in controlling emissions of acid gases, particulates, and heavy metals. The same appears true of wet scrubbers, which also appear to provide the basis for optimum treatment/segregation and management of collected wastes. Evidence that a spray dryer absorber followed by a fabric filter baghouse, operating at a flue gas temperature of 120-140°C (248-284°F), provides efficient removal of heavy metals has been reported. Recent developments, however, suggest that a wet scrubber with additional subcooling of the flue gas may provide optimum control of emissions of some heavy metals and specifically that of gaseous mercury (Hg). The most recent trends in NOx removal point to optimal use of dry non-catalytic techniques recently demonstrated in Switzerland and West Germany.

At the same time, retrofit control systems being added at existing municipal solid waste incineration plants in Sweden to gain compliance with the new (1987) emission limits for PIC's are of interest in viewing the practical application of advanced technologies available in Europe. With seven plants deadlined for compliance in 1988, five in 1989 and six in 1990, nine have installed or made decisions on use/selection of new retrofit gas cleaning facilities (20). Five system are dry type with fabric filters, two utilize wet scrubbing with condensation type treatment and two of the plants will be retired in 1992.

Hazardous Waste

Hospital furnaces have been shown in Sweden (13) to emit dioxins in large amounts. With the aim of developing a program for reducing such emissions from the burning of hospital waste, the following strategies have been established:

- Measurement of dioxin emissions from a number of furnaces, to determine typical characteristics as a function of furnace design and operating conditions
- Comparative study of technological methods for reducing emissions including improvements in combustion and flue-gas cleaning
- Investigation of sending waste to regional incinerators, thus eliminating the need for small furnaces that are only operated intermittently
- A study of sterilization techniques to converting hazardous waste to a condition comparable to ordinary household waste, so reducing the need for use of dedicated hospital incinerators.

EMERGING TECHNOLOGIES

Major MWC technological investigations are going forward in West Germany in the fields of:

- Combined combustion of MW and sewage sludge

- Use of pyrolysis techniques
- Improved waste preparation means
- Recycling of incinerator residues.

The greatest focus and development is reported (1) to be in the technology of dioxin and furan formation and emission. An extensive testing program has shown that there is a very complex distribution of these highly toxic substantces among the various stream flows in a MWC plant. For example, slag generally contains no more than 1 ppb of each of dioxins and furans and bottom ash has up to 10 ppb of each. But fly ash has up to 100 ppb dioxins with up to 1,000 ppb furans, and stack dust emission has up to 1,000 ppb dioxins with up to 10,000 ppb furans. Moreover, measurements show broad scatter indicating that a variety of operational parameters are highly significant. From plant to plant the concentration of individual dioxin and furan species in cleaned stack gas may be as low as 1 to 5 ng/m^3 or as high as 20 to 60 ng/m^3. Moreover, fly ash collected in ESP's from plant to plant may have concentrations of individual dioxin and furan species as low as 0.2 to 2.0 ng/g or as high as 10 to 50 ng/g.

Based on a comprehensive investigative program begun in West Germany in 1986 it has been determined that a considerable proportion of the dioxin formation takes place in the relatively cold post-combustion zones of the MWC plant (e.g. heat exchangers, ducts, dust seperator). However, this doesn't negate the importance of high temperature and long flue gas residence time in the combustion zone. It has been determined that the total destruction of so called "dioxin precursor-compounds" in the combustion zone by high temperature treatment impedes dioxin/furan formation in the downstream zones. Only if there are significant traces left will cold dioxin/furan synthesis be found to occur on the surface of catalytically active dust in the cold zone.

Strategic counter-measures for mitigation of PICs have accordingly been devised in West Germany as follows:

- Minimization of degradable precursor-compounds in the waste (e.g. prohibition of PCP)
- Maintaining of temperature, sufficient oxygen, long residence time, good turbulence in the combustion zone
- Prevention of cold walls in the boiler
- Frequent soot blowing, frequent cleaning of the ESP (i.e. minimization of the residence time of dust in the post-combustion zones)
- Efficient, low-temperature (< 200°C, 392°F) separation of fine dust particulates
- Low excess-air operation (without burnout problems) that prevents dioxin formation.

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TABLE 1. MUNICIPAL SOLID WASTE INCINERATOR EMISSION LIMITS IN WESTERN EUROPE

	West Germany[a]	Sweden	Austria	Switzerland
Solid Particulate Matter, gr/dscf (mg/Nm^3)	0.012 (30)	0.008[d] (20)	0.01[f] (25)	0.02 (50)
Carbon Monoxide, ppm	80	100	—	—
Hydrogen Chloride (inorganic)	31 ppmv (50 mg/Nm^3)	63 ppmv (100 mg/Nm^3)	19 ppmv (30 mg/Nm^3)	19 ppmv (30 mg/Nm^3)
Hydrogen Fluoride (inorganic)	2 mg/Nm^3	—	0.7 mg/Nm^3	5 mg/Nm^3
Sulfur Dioxide	35 ppmv (100 mg/Nm^3)	New SO_2 limits reduce all acids significantly[e]	35 ppmv[g] (100 mg/Nm^3)	175 ppmv (500 mg/Nm^3,) incl. NO_x
Nitrogen Oxides as NO_2	500 mg/Nm^3	—	100 mg/Nm^3	See SO_2
Dioxins and furans, measured as 2.3.7.8 - tetrachlorodibenzo-p-dioxins (TCDD)	b	Existing plants: 0.5-2.0 ng/Nm^3 New plants: 0.1 ng/Nm^3	0.1 ng/Nm^3	—
Total Organics, Common, Class 1, mg/Nm^3	20	—	20	—
Mercury + Cadmium + Thallium, mg/Nm^3	0.2[c]	0.08 (Hg only)	0.1[h]	0.2
Gas Correction	11% O_2 dry	10% CO_2 dry	11% O_2 dry	11% O_2 dry

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[a] Installation of additional burners and of a post combustion area with temperatures of at least 800° (1472°F) is additionally required.

[b] Emission of carcinogenics (e.g. Cr, VI, Ni, benzo(a)pyren, dibenz(a,h)anthracen) and highly hazardous compounds (e.g. dioxins, furans) must be minimized.

[c] Not including vapors: nominal limit for mercury particulate plus vapor is 0.1 mg/Nm^3.

[d] This limit and those below are from the Swedish Environmental Protection Board's "Temporary Emission Goals", July 1986.

[e] See GMTCBR 255-1 for limits.

[f] 20 mg/Nm^3 for new plants of capacity 0.75 to 15 metric ton/hr; 15 mg/Nm^3 above this range.

[g] 50 mg/Nm^3 for new plants of greater than 15 metric ton/hr capacity.

[h] 0.05 mg/Nm^3 for new plants of greater than 15 metric ton/hr capacity.

TABLE 2. ESTIMATED DIOXIN EMISSIONS IN SWEDEN

Identified Sources	TCDD Equiv., g/yr
Incinerators[a](flue-gases)	50-100
Motor vehicles(exhaust gases)	5-15
Iron, steel, and nonferrous metal works(flue-gases)	50-150
Pulp and paper mills(flue-gases)	4-6
Ditto(effluents)	15-30
Coal-fired power plants(flue-gases)	1
Hospital furnaces(flue-gases)	10
Burning of hazardous waste(flue gases)	2-6

[a] 1985, prior to new standards, 200-300 g/yr; from 1992 about 10 g/yr.

7C-62

A NEW CONCEPT: SEPARATELY-FIRED SUPERHEATERS FOR WASTE-TO-ENERGY PLANTS

BY

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ABSTRACT

This paper proposes a concept of improving the efficiency and economics of municipal solid waste (MSW) fired energy recovery plants by use of natural gas. The concept consists of:

- removing the superheater from the mass-fired boiler to eliminate corrosion problems commonly associated with this equipment
- modifying the mass-fired boiler to generate saturated steam at 1800 psi.
- providing a separate natural gas-fired superheater to raise steam temperature to 1000°F

The concept was analyzed in a study by Camp Dresser and McKee, Inc. assisted by R. E. Sommerlad, Consultant, based on a general concept proposed by the New York State Energy Research and Development Authority.

When applied to a 1200 tpd MSW plant, the following conditions may be expected with gas purchased at $3.50 per million Btu and electric power sold at $0.06 per kWh:

- reduction in tipping fees $4.30 per ton
- reduction in total yearly costs $1.7 million
- incremental capital cost $460 per kW
- incremental capital cost $6,550 per tpd
- additional electricity sales 122,000 MWh per year

These incremental values apply in comparison to a similarly sized resource recovery plant designed on steam conditions of 900psig/830°F.

A dispersion analysis to determine the effect of combining the stack gases from the MSW-fired boiler and the separately-fired superheater shows a reduction in ground level concentration of 12.5 percent when compared with the MSW-fired boiler alone. The removal of "low-end" heating surface (e.g., air heater) to increase the stack temperature reduces ground level concentration by 29 percent, albeit at increased fuel cost. In any case, the predicted air quality impacts for either the individual or merged flue gas streams are well within environmental standards and it is concluded that merging is not required.

INTRODUCTION

The corrosive nature of combustion gases and ash deposits on superheater tubes, when burning municipal solid waste (MSW), limits the production of high temperature superheated steam that is desirable for efficient electric power generation with steam turbines. While utility boilers produce 2,600psig/1000°F or higher quality superheated steam, final steam conditions of MSW-fired boilers typically peaked at 900psig/830°F, limiting electric power generating potential. Lower temperature steam for MSW-fired boilers is selected to minimize tube metal deposits, fireside corrosion, and slagging, thereby reducing maintenance and downtime. This compromise, however, is made at some sacrifice of electric power generation.

The approach discussed in this paper is to separate the superheater and steam raising sections of the MSW incinerator (i.e., use a separately-fired superheater) to optimize the trade-off between high steam quality and low maintenance and downtime. The objective is to produce high pressure saturated steam in the MSW boiler and superheat this saturated steam up to about 1,000°F in a natural gas-fired superheater. Based on PURPA limitations natural gas heat input would be less than 25 percent of total heat input, depending on the pressure of saturated steam from the MSW boiler and final superheater steam conditions. The resulting benefits include increased and more efficient electric power production due to higher quality steam conditions, and reduced MSW boiler maintenance and downtime as a result of lower tube metal temperatures.

A concern resulting from operating at higher steam pressures is the impact on the integrity of the furnace walls and other convection surfaces. Moving the superheater to an environment where the surface is in contact with flue gas free of acid gases and particulate matter eliminates any negative impact on superheat surface. At steam pressures of 900 psig, the saturation temperature is 533°F and the average fireside tube metal temperature is approximately 580°F. Typically, the threshold for the breakdown of tube integrity begins at a metal temperature of 800°F. At 1,800 psig, the saturation temperature is 620°F and the average fireside tube metal temperature is 670°F which is well below the breakdown fluid temperature of 800°F.

This reasoning, supported by experience both in the U.S. and abroad, indicates that high pressure steam (1,800 psi and above) can be generated in an MSW-fired environment while maintaining tube surface integrity. If power generation is to be maximized, higher steam temperatures are required. This leads to the concept of producing such temperatures with fossil fuels, such as natural gas in a separately-fired superheater. The resulting design concept consists of an MSW-fired boiler (or evaporator) generating saturated steam at the highest acceptable pressure and then superheating with natural gas.

BOILER DESIGN

Past practices of steam turbine applications by the power industry indicated that steam conditions range in capacity as shown in Table 1.

TABLE 1

Typical Power Generation Steam Conditions and Size Ranges

Steam Conditions [Pressure (psig)/ SH(°F)/RH(°F)]*	Saturation Temp.(°F)	Associated Size Range (MW)	Steam Flow (10^3 lb/hr)
2400/1000/1000	676	215-500	1551-3481
1800/1000/1000	622	85-215	669-1527
1250/950/----**	534	40-85	415-785
850/900/----**	527	<40	<405

*Superheat Temperature/Reheat Temperature
**Nonreheat systems

For a 1,200 tpd MSW-plant a conventional 900psig/830°F cycle will generate about 34 MW. A survey of the turbine manufacturers indicated that turbines are available at steam pressures up to 2000 psig and temperatures of 1000°F in the size range of interest. A pressure of 1800 psig and a temperature of 1000°F were selected for detailed evaluation based on the wide application of these conditions by the utility industry. Both reheat and non-reheat cycles were considered. At 1,200 tpd the 1800psig/1000°F cycle results in a turbine/generator capacity of 50.9 MW, or a capacity increase of almost 50 percent. Examples of the above relationships are shown on a fuel map in Figure 1.

With the aid of our in-house computerized cycle analysis program, a variety of feedwater heating systems were evaluated. The system configurations analyzed ranged from the typical utility system comprised of six or seven feedwater heaters and an air heater, to a typical MSW system with only two or three feedwater heaters and no air heater. An extraction steam system consisting of a simple feedwater heater and deaerator was chosen on the basis of thermodynamic compatibility and present day MSW waste-to-energy practice. The entire hybrid thermal power cycle is shown in Figure 2.

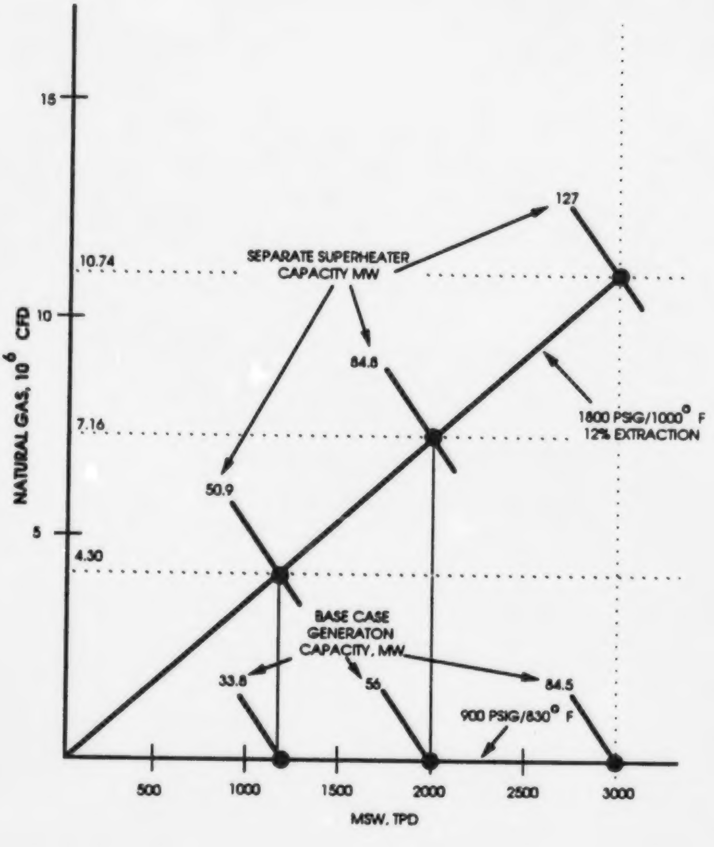


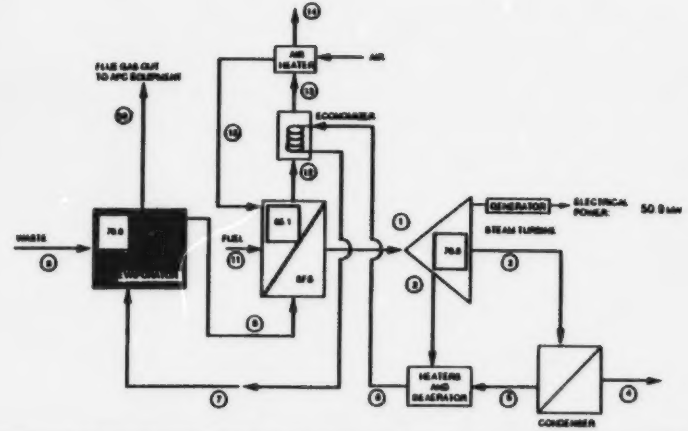
FIGURE 1
FUEL MAP FOR SEPARATELY-FIRED SUPERHEATER

HEAT IN	HEAT OUT
500.0	145.6
180.7	27.0
	334.3
	173.0
680.7	680.7

CYCLE EFF'Y = $\frac{173.0}{680.7}$

= 25.4%

☐ EFFICIENCY, %



WASTE
⑨
70.0
FLUE GAS OUT TO APC EQUIPMENT
⑩
FUEL
⑪
85.1
SFS
⑧
⑦
⑫
ECONOMIZER
⑬
AIR HEATER
AIR
⑭
⑮
①
②
79.0
GENERATOR
ELECTRICAL POWER
50.8 MW
STEAM TURBINE
③
HEATERS AND DEAERATOR
⑥
⑤
CONDENSER
④

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
								B O I L E R			S E P A R A T E L Y - F I R E D S U P E R H E A T E R				
USE	THROTTLE STEAM	EXTRACTION FLOW	TURBINE EXHAUST	CONDENSER COOLING WATER	CONDENSATE TO HEATERS	CONDENSATE FROM HEATERS	PREHEATED FEEDWATER TO BOILER	SATURATED STEAM	FUEL/ WASTE	FLUE GAS EXHAUST	FUEL	FLUE GAS TO ECONOMIZER	FLUE GAS TO AIR HEATER	FLUE GAS EXHAUST	PREHEATED AIR TO SFS
MASS FLOW (M LB/HR)	402.8	48.3	354.5		354.3	402.8	402.8	402.8							
HEAT FLOW (MM BTU/HR)	588.3	58.7	363.7	334.3	29.4	88.1	109.2	483.7	500	145.6	180.7	70.0	49.0	27.0	22.0
PRESSURE (PSIG)	1800		2.0					1900							
TEMPERATURE (°F)	1000		115		115	251	303	622							
ENTHALPY (BTU/LB) OR HEAT VALUE	1460	1214	1026		83	219	271	1151							

* Includes Fig. A

FIGURE 2
HYBRID THERMAL POWER CYCLE SCHEMATIC

STEAM CONDITIONS
PRESS. 1800 PSIG TEMP 1000 DEGREES F
EXCESS 12 %

Although several 1,000 tpd and at least two 1,300 tpd furnaces have been built and successfully operated, only multiple furnace units were considered:

- 2 - 600 tpd
- 3 - 400 tpd
- 4 - 300 tpd

In choosing 3 - 400 tpd units as indicative of present MSW plant sizing, it was decided that one steam turbine would be utilized for three units without thermal penalty. Assuming one superheater for each boiler, a flow chart of the system is as shown in Figure 3.

In preparing the design of the new system it became apparent that a conventional MSW water-wall unit (with an integral superheater) and a high pressure boiler (evaporator) without any superheat surface, have essentially the same components (Table 2). To illustrate this comparison, it was decided to modify the design of an existing 400 tpd unit (City of Commerce, California) "on paper" to reflect a new higher pressure boiler (evaporator).

The re-design incorporates the modifications shown in Table 2 including: increase in the thickness of pressure parts to accommodate an increase in steam pressure, removal of the superheater, replacement of the boiler bank, and modification of the economizer. The screen tubes and the first bank of evaporator tubes were kept the same. The second bank of evaporator tubes was expanded from 4 to 6 rows. A third and fourth bank of evaporator tubes similar to the second tank were added. The economizer plan dimensions were kept the same, but additional rows of tubes were added to accommodate the increase in duty resulting from an increase in steam flow.

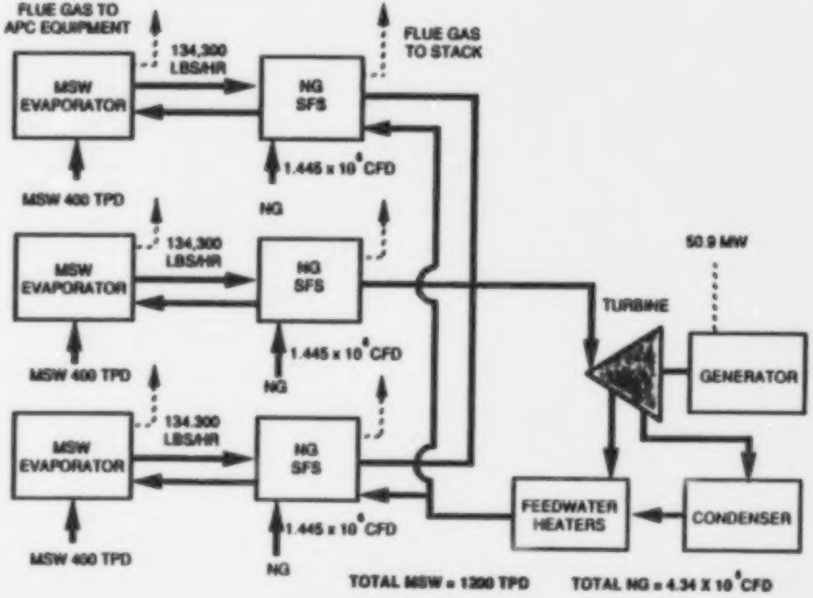

FLUE GAS TO APC EQUIPMENT
134,300 LBS/HR
FLUE GAS TO STACK
MSW EVAPORATOR
NG SFS
MSW 400 TPD
1.445 x 10^6 CFD
NG
134,300 LBS/HR
MSW EVAPORATOR
NG SFS
50.9 MW
TURBINE
MSW 400 TPD
1.445 x 10^6 CFD
NG
GENERATOR
134,300 LBS/HR
MSW EVAPORATOR
NG SFS
FEEDWATER HEATERS
CONDENSER
1.445 x 10^6 CFD
MSW 400 TPD
NG
TOTAL MSW = 1200 TPD
TOTAL NG = 4.34 X 10^6 CFD

FIGURE 3
HYBRID CASE B-1 - FLOW CHART
1800 PSIG 1000 DEGREES F

Table 2

BOILER COMPONENTS GUIDELINES

Components	Modifications
Water Cooled Chute	None
Feeder	None
Grate	None
Underfire Air System	None
Overfire Air System	None
Residue-Siftings Removal	None
Furnace	None
Pressure Parts	Increase tube and header thicknesses for increased pressure
Slag Screen	None
Superheater	Remove
Boiler Bank	Modify surface area
Economizer	Modify surface area
Steam Drum	Modify as required
Feeder Tubes	Modify as required
Air Pollution Control System	None

DESIGN OF NATURAL GAS-FIRED SUPERHEATER

The following factors must be considered in evaluating the design of a natural gas-fired superheater design cycle:

- Feedwater temperature
- Steam pressure and temperature
- MSW boiler capacity

Regenerative feedwater heating is an effective means for improving the efficiency of the Rankine cycle; however, even the most modern MSW plants offer few incentives to maximize regenerative feedwater heating. Typically, regenerative heating amounts to 0-20 percent extraction, usually arranged in two to three feedwater heaters. This results in a feedwater temperature of about 300°F which is compatible with a flue gas exit temperature of 425°F to 450°F.

Once the steam pressure and number of units were chosen, the superheater design could proceed and the state points established. The excess air for the natural gas burner was set at 10 percent. The furnace circuitry (radiant superheater) consists of 3 in. O.D. tubes arranged on 5-1/2 in. centers on a furnace tube pitch diameter of 14 ft-0 in. The diameter of the furnace refractory wall is 15 ft-0 in. The convection superheater comprises horizontal tubes in a conventional tube bundle array and is mounted above the furnace. Because of the relatively high temperature of the saturated vapor, the flow is first to the radiant section and then to the convection section of the superheater (Figure 4).

If the only heat transfer surface in the separately-fired superheater was for heating saturated vapor from 620 to 1000°F, then the flue gas would be exhausted at a temperature of about 720°F. The option of additional heat recovery was analyzed and found to have technical and economic merit.

Based on the cycle analysis shown in Figure 2 (a 1800psig/ 1,000°F cycle with 12 percent extraction), the feedwater leaving the feedwater heaters is directed to an economizer in the natural gas-fired superheater; thus decreasing the flue gas temperature and the feedwater temperature. The feedwater is then directed to the economizer in the MSW-fired boiler as in a conventional plant. In terms of flow, the connection of feedwater from the feedwater heaters, to the "pre-economizer" in the natural gas-fired superheater, to the economizer in the MSW-fired boiler, is a series path. The "pre-economizer" is, in effect, another

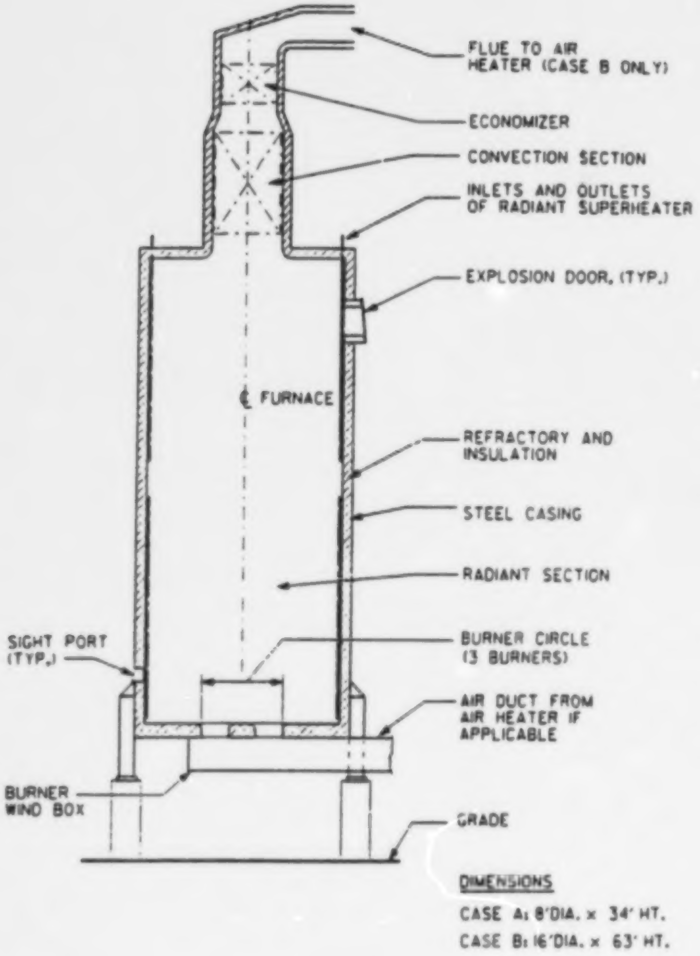

FLUE TO AIR HEATER (CASE B ONLY)
ECONOMIZER
CONVECTION SECTION
INLETS AND OUTLETS OF RADIANT SUPERHEATER
EXPLOSION DOOR, (TYP.)
℄ FURNACE
REFRACTORY AND INSULATION
STEEL CASING
RADIANT SECTION
SIGHT PORT (TYP.)
BURNER CIRCLE (3 BURNERS)
AIR DUCT FROM AIR HEATER IF APPLICABLE
BURNER WIND BOX
GRADE
DIMENSIONS
CASE A: 8'DIA. × 34' HT.
CASE B: 16'DIA. × 63' HT.

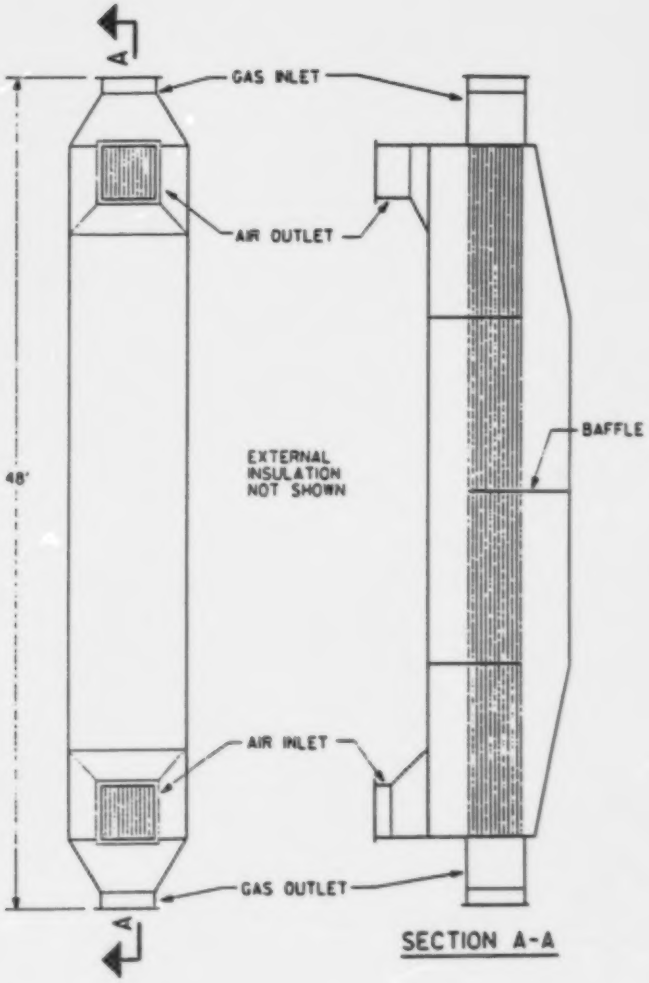
FIGURE 4
SECTIONAL VIEW OF SUPERHEATER

feedwater heater, except that its source is flue gas instead of extraction steam.

As also shown in Figure 2, the temperature leaving the "pre-economizer" in the natural gas-fired superheater is sufficient to include and justify an air heater (Figure 5). A flue gas exit temperature of 285°F is compatible with accepted practice. Thus, the separately-fired superheater with 10 percent excess air and an outlet temperature of 285°F corresponds to a thermal efficiency of 85.1 percent.

Based on the above considerations, the following design characteristics were chosen:

- Heat Input - 180.7 x 10^6 Btu/hr
- Furnace exit temperature - 1,800°F
- Air preheat temperature - 500°F
- Parallel flow convection superheater
- Parallel flow economizer


A
GAS INLET
AIR OUTLET
BAFFLE
EXTERNAL INSULATION NOT SHOWN
48'
AIR INLET
GAS OUTLET
A
SECTION A-A

MATERIAL: CARBON STEEL

FIGURE 5
AIR HEATER

FACILITY CONSIDERATIONS

The controlling spatial dimensions for the facility, including the refuse storage pit, boilers, superheaters, dry scrubbers, and fabric filters, are:

- Pit width
- Train length (pit to stack)

The three boilers with their air pollution control equipment, when arranged side-by-side, provide ample room for the turbine and three superheater systems as shown in Figure 6. The reduction in the boiler space required due to removal of the superheater is equivalent to the space taken up by the separately-fired superheaters. Therefore the net effect on the overall spacial requirements for the facility is expected to be negligible.

Table 3 illustrates the performance of the hybrid plant. It shows an incremental heat rate of 10,559 Btu/kWh, representing the additional (incremental) export power of 16.4 MW produced by the natural gas fired superheater.

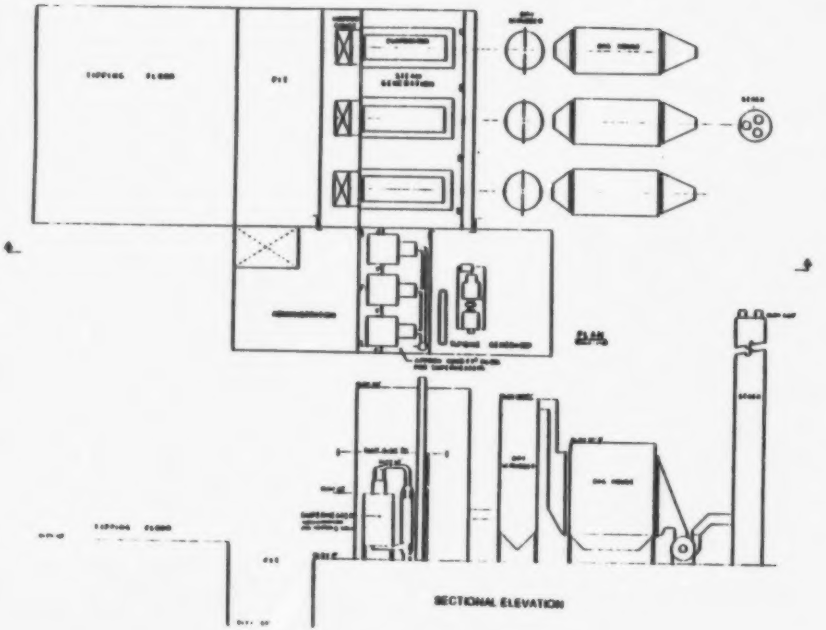

TIPPING FLOOR
PIT
STACK
PLAN
TIPPING FLOOR
PIT
SECTIONAL ELEVATION

FIGURE 6
FACILITY LAYOUT

Table 3

COMPARATIVE CYCLE PERFORMANCE

	Conventional Cycle 900psig/830°F	Hybrid Cycle 1800psig/1000°F
Refuse Capacity, tpd	1,200	1,200
Gross Power, MW	33.8	50.9
Export Power, MW	29.8	46.2
Incremental Power, MW	-	16.4
Fuel Use, 10^6 Btu/hr	-	180.7
Station Heat Rate, Btu/kWh (gross)	14,782	13,363
Station Heat Rate, Btu/KWh (Net)	16,778	14,734
Incremental Heat Rate, Btu/kWh	-	10,559

ECONOMICS

The cost estimates for the boiler and superheater are shown in Tables 4 and 5, respectively. As indicated in the tables, the cost of the boiler as a result of the re-design is no more costly than the base case. The superheaters' cost is approximately $1 million each.

To evaluate the incremental economics due to the use of the separately-fired superheater it is assumed that no changes are necessary to the "front end" of the plant (tipping floor, pit, cranes) or the "back-end" (air pollution control equipment). An evaluation of the floor space indicates that no additional building space is needed for the separately-fired superheaters. The space saved by shortening the boiler (due to removal of the integral superheater) is balanced by the space requirements of the separately fired superheaters (see Figure 6).

Except for the use of natural gas, no additional operating costs are anticipated as a result of this design concept. The expected reductions in boiler maintenance cost due to expected decreases in corrosion have not been taken into account, resulting in a more conservative comparison.

Amortization of the incremental investment is based on borrowed funds at 10 percent over twenty years.

Gas costs are taken between $2.00 and $6.00 per million Btu and revenue for power sales are analyzed for a range of $0.04 per KWh to $0.08 per KWh.

With gas at $3.50 per million Btu and electricity sales at $0.06/KWh, a reduction in the tip fee of $4.51 per ton may be expected. This corresponds to an increase in income (or a reduction in operating cost) of $1.7 million per year for an investment of $7.9 million. The average tipping fee for large MSW energy recovery facilities has been reported at $50.00 per ton at present. The hybrid plant, therefore, offers a cost reduction of 9.0 percent for the conditions considered above.

The more generalized case for the range of fuel and electricity costs considered is shown in Figure 7. Even with the conservative assumptions noted, the hybrid concept offers a reduction of the tipping fee over a wide range of fuel costs depending on the price at which power can be sold. For instance, if natural gas can be purchased at $3.00 per million Btu, a positive revenue stream can be expected at power sales over $0.04

Table 4

COST ESTIMATE FOR BOILERS - BASE AND HYBRID CASES

This cost estimate is a cost comparison of a conventional unit with integral superheater (referred to as Base) to the 1800 psig saturated steam boiler (referred to as Hybrid), including the various pressure parts, headers, casing, structural steel, etc., but excluding the mass burn grate (stoker, drives, etc.), water cooled chute and flues and ducts, air heater, fans and motors, and residue and fly ash removal systems. The excluded parts are essentially the same for both cases. This cost estimate is for one of three 400-tpd units and consists of the following items:

- Pressure Parts
  - Upper Drum
  - Drum Internals
  - Lower Drum*
  - Furnace
  - Evaporator
  - Wing Walls
  - Boiler Bank*
  - Primary Superheater*
  - Secondary Superheater*
  - Economizer
  - Seals
  - Boiler Sections
- Other Parts
  - Buckstays
  - Hangers and Springs
  - Structural Steel
  - Insulation
  - Setting
  - Sheathing (Aluminum)
  - Casing
  - Framing
  - Soot Blowers
  - Platforms
  - Valves and Fittings
  - Doors
  - Misc. Drain and Vent Piping
  - Reinjection Piping and Valves
  - Gas Burner

	Base	Hybrid	Diff.
Weight, lbs.	1,353,000	1,459,000	negligible
Cost	$2,161,000	$2,170,000	negligible

---

* Eliminated in Hybrid

Table 5

COST ESTIMATE FOR SEPARATELY-FIRED SUPERHEATERS

This cost estimate is based on the design of one of three units and consists of the following items:

- Pressure Parts
  - Furnace
  - Convection Superheater
  - Economizer
  - Air Heater
- Other Parts
  - Hangers and Springs
  - Support Steel
  - Insulation
  - Setting
  - Casing
  - Gas Burners
  - Platforms and Ladders
  - Valves and Fittings
  - Doors
  - Burner Windbox
  - Misc. Piping
  - Fans and Motors
  - Flues and Ducts
  - High Pressure Steam Piping (Superheater to Turbine)
  - Stack

Superheater (1 of 3 Units)

Weight, lbs.	418,500
Total Cost for 1 Unit	$1,089,000
Total Cost for 3 Units	$3,267,000

TABLE 6: INCREMENTAL COST FOR SEPARATELY-FIRED SUPERHEATER

STEAM CONDITONS - BASE: 900PSIG/830F HYBRID: 1800PSIG/1000F
REFUSE CAPACITY - 1200 TPD

INCREMENTAL CAPITAL COSTS:

	UNIT COST	# OF UNITS	INCR. COST
FURNACE,FEEDER,ASH REMOVAL	0	3	0
SAT STEAM BOILER	0	3	0
AIR POLLUTION CONTROL	0	3	0
SEP-FIRED SPRHTR incl.install'n	$1,089,000	3	$3,267,000
F/W PUMP,TRTMNT,CONTROLS, PIPING	$250,000	3	$750,000
INSTALLATION,SHIPPING	$0	3	$0
ADDITIONAL FLOOR SPACE, $/SQFT	$150	0	$0
INCR TURB-GEN COSTS, $/KW,KW	$150	17,100	$2,565,000
CONDENSER,COOLING TOWER ETC	50.0%		$1,282,500
		TOTALS:	$7,864,500

INCREMENTAL O&M COSTS:

	HYBRID	BASE
AVAILABILITY:	0.85	0.85
OPERATING HOURS:	7446	7446
YEARLY WASTE TONNAGE	372,300	372,300

LABOR				$0
AIR POLLUTION				$0
POWER COST OF SFS, $/KWH,MWH		$0.06	*	$0
CAPITAL REC @ 11%,20 YRS		0.119		$935,876
GAS USE, $/MMB, MMBH		$3.50	180.7	$4,709,223
				$5,645,098
INCR ELEC REVENUE	$/KWH,MWH/YR	$0.06	122,089	$7,325,340
	INCREMENTAL INCOME(OUTGO)			$1,680,242
	REDUCTION(INCREASE)PER TON			$4.51

* INCL IN INCR ELEC REVENUE

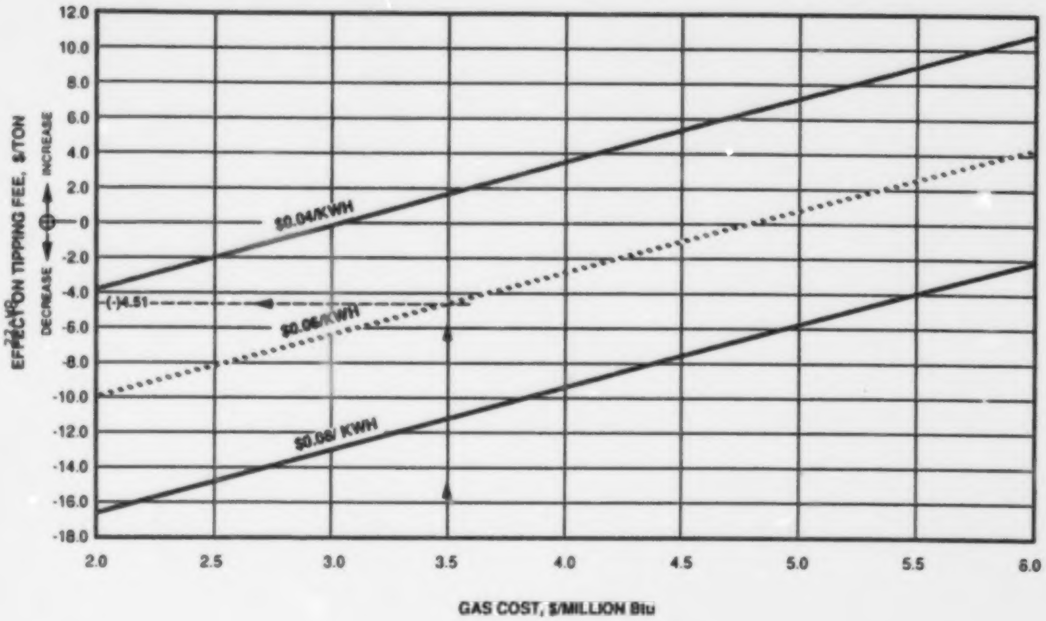


FIGURE 7
EFFECT OF FUEL COST AND ELECTRIC REVENUE

per kWh. With gas at $4.80 per million Btu, export power must be sold at $0.06 per kWh to justify this concept. With electric revenue at $0.08 per KWh, gas costs can be $6.00 per million Btu and still justify this concept.

SENSITIVITY ANALYSIS

A sensitivity analysis was carried out using as the reference case, the 1,800psig/1000°F condition with gas at $3.50 per million Btu and electric power revenue at $0.06 per kWh which produced a reduction in tipping fee (the SFS benefit) of $4.51 per ton. Capital cost, fuel cost, and electric revenue were varied by 10 percent. In addition, the effect of varying steam conditions and turbine efficiencies were examined. The results of this analysis, presented in Table 7 may be summarized as follows:

- A 10 percent increase in capital cost results in a reduction in the SFS benefit of 5.6 percent. In other words, the tip fee benefit declines from $4.51 per ton of the reference case is reduced to $4.26 per ton if capital cost increases by 10 percent.
- An increase of 10 percent in the fuel cost reduces the SFS benefits by 28.0 percent to $3.25 per ton, whereas an increase of power revenue by 10 percent increases the benefit by 43.6 percent (equivalent to a reduction in tip fee of $6.48 per ton).
- Similarly, if turbine efficiency can be raised from the assumed 78 percent by two percentage points due to improved design as steam conditions, the SFS benefit would rise by 31.9 percent over the reference case. This is equal to a reduction in tip fee of $5.95 per ton bringing the average tipping fee from $50.00 per ton to $44.05 per ton.

The sensitivity analysis indicates that the cost of fuel relative to electric revenue exerts a major impact· on the ultimate financial return.

TABLE 7 - SENSITIVITY ANALYSIS

VARIABLE COLUMN #	REF CASE	SFS EFF'Y (1)	TURB EFF'Y (2)
STEAM CONDITIONS	1800/1000	1800/1000	1800/1000
SFS EFFICIENCY	85.10%	83.40%	85.10%
TURB EFFICIENCY	78.00%	78.00%	80.00%
FUEL, MMBH	180.7	184.3	180.7
EXP POWER, MW	46.2	46.2	47.4
INCR EXPORT, MW	16.4	16.4	17.6

	REFERENCE CASE			VARIABLES					
	UNIT COST	UNITS *	ANNUAL COSTS	CAPITAL COSTS	FUEL COSTS	ELECTRIC REVENUE	SFS EFF'Y 1800/1000 (1)	TURB EFF'Y 1800/1000 (2)	AVAIL'Y 90.00% (4)
COLUMN #:									
CAPITAL RECOVERY	0.119	$7,864,500	($935,876)	10.0%					
FUEL USE	$3.50	180.7	($4,709,223)		10.0%		2.0%	0.0%	5.9%
INCR PWR SALES	$0.06	$122,089	$7,325,340			10.0%	0.0%	7.3%	5.9%
NET YEARLY REDUC'N(INCR)			$1,680,242	$1,586,654	$1,209,320	$2,412,776	$1,586,422	$2,216,242	$1,834,131
REDUC'N(INCR) IN TIP FEE			$4.51	$4.26	$3.25	$6.48	$4.26	$5.95	$4.93
CHANGE FROM REFERENCE			0.0%	-5.6%	-28.0%	43.6%	-5.6%	31.9%	9.2%
ESTIMATED TIPPING FEE **			$45.49	$45.74	$46.75	$43.52	$45.74	$44.05	$45.07
CHANGE FROM BASE FEE**			-9.0%	-8.5%	-6.5%	-13.0%	-8.5%	-11.9%	-9.9%

* REFER TO TABLE 6
** BASIC TIPPING FEE = $50.00 PER TON

ENVIRONMENTAL CONSIDERATIONS

Natural gas combustion results in a low emissions profile with little or no particulate and acid gas emissions. For a furnace using natural gas, the only emission of concern is NO_x which can be kept at acceptable values. Federal regulations do not apply in this case because the heat input is less than 100 million Btu/h. Carbon monoxide, sulfur oxides and total hydrocarbon emissions are minimal.

Merging of fuel gas streams is sometimes used to gain emission control advantages as seen in the Munich South Plant. Two units at the Munich South Plant were designed for 1,000 tpd in a separate refuse-fired economizer. The boiler was designed for the future use of pulverized coal. Flue gas from the MSW-fired economizer was blended with flue gas from the boiler and passed through the air pollution control system which at the time (circa 1970) consisted of a electrostatic precipitator (ESP). The ESP was designed and installed for particulate matter from coal and MSW combustion, even though the boiler was initially fired with natural gas. Coal pulverizers were never installed. The ESP then handled the MSW-fired flue gas laden with particulate and the natural gas-fired flue gas with no particulate matter. Recently, this system was revamped with a dry scrubber to treat only the MSW-fired flue gas.

The merging flue gases from the MSW-fired boiler and separately-fired superheater increases the flow rate of the exhaust gases leaving the stack. The change in flue gas temperature would depend on where the flue gases were merged. If the merging took place before the air pollution control systems, the stack temperature of the flue gas would be regulated by the acid gas scrubber. However, the efficiency of the scrubber could be affected and the gas volume would be increased by 21 percent with a corresponding increase in cost of the subsequent air pollution control devices. If the flue gases are merged after the air pollution control devices, the gases from the separately-fired superheater would increase the temperature of the flue gases from the MSW-fired boiler. The amount of the increase in temperature would depend on whether an air heater was used.

To investigate the effect of merged flue gas streams, a dispersion analysis was made for the 1,200 tpd hybrid facility with and without combined flue gas streams. The merging of the flue gas streams was assumed to occur after the air pollution control device because of its lesser impact in capital costs. Operation of the separately-fired superheater was evaluated with and without the use of an air heater.

The results of the modeling analysis are normalized in that the emission rate used in the modeling is one gram per second. The predicted normalized concentration at the point of maximum impact are summarized in Table 8. To predict the maximum ground level concentration for an individual pollutant, the predicted concentration must be multiplied by the estimated emission rate in grams per second for that pollutant.

For the reference case, the normalized impacts for the natural gas-fired superheater are higher than the MSW-fired boiler because the stack height and volumetric flow rate are lower. In the first case, the natural gas-fired unit is operated with an air heater. The maximum normalized impacts were 1.159 ug/m^3 or 12.5 percent less than impacts for the MSW-fired boiler alone. In the second case, the natural fired unit is operated without the air preheater. The maximum normalized impacts were 0.942 to 0.939 ug/m^3 or 28.9 to 29.1 percent less than impacts from the MSW-fired boiler alone. This comparison of normalized impacts represents the enhanced plume dispersion from the greater plume rise and buoyancy flux associated with the merged flue gas streams.

For most non-criteria pollutants including trace metal and organic compounds, a corresponding reduction in the ground level concentration will be observed. For the criteria pollutants, the separately-fired superheater contributes to the pollutant emissions for the merged flue gas streams. The increased plume dispersion offsets the increase in emissions associated with the operation of the separately-fired superheater. The predicted air quality impacts for either the individual or merged flue gas streams are below the New York State Ambient Air Quality Standards and Acceptable Ambient Levels.

Table 8

SUMMARY OF ISCST MODELING ANALYSIS OF INDIVIDUAL AND MERGED FLUE GAS STREAMS

	Maximum Normalized Ground Level Impacts[1] ($\mu g/m^3$)	Distance to Point of Maximum Impact (km)
Reference Case		
Refuse Fired Boiler	1.32	0.44
Natural Gas Superheater with Air Preheater	4.65	0.31
Natural Gas Superheater without Air Preheater	2.08	0.37
Case No. 1 - Combined Flue Gas Streams with Air Preheater		
Constant Stack Diameter	1.16	0.44
Constant Exit Gas Velocity	1.16	0.44
Case No. 2 - Combined Flue Gas Stream without Air Preheater		
Constant Stack Diameter	0.942	0.50
Constant Exit Gas Velocity	0.939	0.50

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[1] Based on an emission rate of 1 g/sec

CONCLUSIONS

In summary, the study indicates that a seperate gas-fired superheater when included in a waste-to-energy cycle, improves not only efficiency but also plant economics due to the higher steam pressures and temperatures possible. While conventional waste-to-energy plants have recently been designed for steam conditions at 900psig/830°F the use of a separately-fired superheater may allow the generation of steam at 1800psig/1000°F. These steam conditions appear to be the highest values for which steam turbines are commercially available in the capacity range considered here (up to 80 MW).

The study proves that reductions in tipping fee of $4.50 per ton may be expected assuming electricity can be sold at $0.06 per kWh and gas purchased at $3.50 per million Btu. A sensitivity analysis shows that, even at lower pressures and temperatures (1250psig/950°F) a tipping fee reduction of $2.98 per ton may still occur. The relative costs of electricity and natural gas appear to exert the greatest influence on the economic viability of this design concept.

An environmental study was made of the effect on ground-level concentrations of pollutants from MSW facilities with a separately-fired superheater. The predicted air quality impacts appear to be well within environmental standards whether the stack gases from the MSW-fired boiler and the gas-fired superheater are combined or not.

As a next step a specific plant design incorporating this concept should be undertaken in order to establish capital and operating costs on a firm basis.

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DESIGN AND OPERATION RESULTS ON MSW FLUIDISED BED INCINERATOR

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ABSTRACT

To burn wastes of high calorific value and to meet stringent environmental requirements, fluidised bed incinerators has advantages over incinerators of other types. In Japan, about one hundred MSW fluidized bed incineration plants are now in operation and under construction, and more fluidised bed incinerators will be constructed in the future.

Comparing with stoker type incinerators, fluidised bed incinerators have the following features:

- Various types of refuse, including low-calorie sludge and high-calorie refuse containing a large amount of plastics, can be burnt separately or in combination.
- Simple construction without internal mechanical moving parts makes maintenance easier.
- Complete combustion and dry ash discharge reduce ash treatment cost and secondary pollution problem by water, and make the life of landfill site longer.
- High heat content makes batch operation easier.

This paper discusses the design and operational data of MSW fluidized bed incinerators currently operating in Japan, including operating conditions and the amount of generated dioxin.

INTRODUCTION

The concept of fluidized beds goes back to the fluidized calcining furnace in 1879. Probably the first fluidized bed to operate on a commercial scale was the Winkler gas generator in 1921. Since then, these units spread all over the world. The rapid development of fluidization technology began in the first half of the 1940s with the development of the fluid catalytic cracking unit. Currently, in addition to the use of fluidization technology for petroleum refining, sulfide ore roasting and limestone calcining, it is also used for burning coal, sewage sludge and municipal solid waste (MSW).

The history of MSW incinerators in Japan began with the construction of an 11.5 t/d batch incinerator in 1900. The first continuous stoker type incinerator, with a capacity of 18 t/d was built in 1960. An MSW incinerator with a power generator first appeared in 1972. The first MSW incineration plant utilizing a fluidized bed was developed in Japan in 1972 (24 t/d). Since then, about one hundred MSW fluidized bed incineration plants are now in operation and under construction. In the last three years, there have been orders for 32 fluidized bed incinerators, while 57 mechanical stoker type incinerators have been ordered for continuous and semi-continuous (8 or 16 hr/day operation) incineration plant. The share of fluidized bed incinerators, however, is steadily increasing.

MSW FLUIDIZED BED INCINERATION SYSTEMS

Figure 1 shows general flow of MSW fluidized bed incineration plant, and Figure 2 shows MSW fluidized bed incinerator.

Ordinary combustible refuse is dumped into a refuse pit from refuse vehicles, and bulky refuse such as carpets, wooden waste, wooden furniture, etc., is cut by a bulky refuse shredder and shifted to the refuse pit.

Refuse in the refuse pit is fed to a refuse feeder by a refuse crane. The refuse feeder feeds refuse to the incinerator uniformly.

Incinerators are constructed of refractory materials and steel walls, and are in the shape of a cylinder or rectangle. They are divided, from top to bottom, into the freeboard, the fluidizing section, the air spray section, and the incombustibles discharging section. The refuse feeding hole, the heatup burner, and the secondary air nozzles are located at midway elevation of the incinerator. The depth of fluidizing sand is about 500 to 1,000 mm at stopped condition.

The fluidized bed is heated and a high temperature is maintained by a heatup burner. When the fluidized bed has reached the specified temperature (higher than 550°C), the heatup burner is cut, and the refuse can be fed in.

Because the fluidized bed temperature is maintained between 650 to 750°C, refuse begins to burn immediately. The refuse passes through the freeboard section which is maintained at 700 to 950°C, where it is burned completely.

Fluidized bed incinerators are normally capable of burning 750 kcal/kg refuse without auxiliary fuel, and if fluidizing air is heated to a high temperature by hot flue gas, 600 kcal/kg refuse can be burned without auxiliary fuel.

Fluidizing air is led to the fluidizing bed through spray nozzles at the bottom of the incinerator by the primary air blower, and secondary air is supplied to the freeboard section by the secondary air blower to ensure complete combustion.

Incombustibles are discharged with the fluidized bed material through a bed material discharger installed in the bottom of the incinerator. They are separated from the fluidized bed material by a separating screen. Small materials are reused as bed material, and the remaining incombustibles are taken out of the system.

Flue gas from the incinerator is cooled to about 300°C by a waste heat boiler or a gas cooler and led to an electrostatic precipitator to remove ash. The purified gas containing dust at less than 50 mg/Nm^3 is then discharged into the atmosphere by an induced draft fan.

Heat of the flue gas is utilized to generate steam in the waste heat boiler, and the steam is used to generate electric power. The power can be used in the incineration plant or sold to an electric power company, and also steam is supplied for district heating outside the plant.

The overall NOx generation is as low as 150 ppm, since a small amount of excess air is supplied to the fluidized bed incinerator. A two-stage combustion system can reduce the NOx content to less than 100 ppm.

Hydrogen chloride generated by the combustion of petrochemical products in the refuse is corrosive and harmful, so HCl removal is very important.

Pit slops are oxidized in the incinerator. The wastewater from the plant is treated and re-used within the plant.

The foul odors in the refuse pit are sucked into the incinerator and decomposed at high temperature. The entire system is designed to prevent foul odors from being emitted into the surrounding atmosphere.

DESIGN CRITERIA OF FLUIDIZED BED INCINERATOR

To design the fluidized bed municipal refuse incinerator, several important factors need to be studied, as itemized below.

Principles of fluidization
Refractory construction
Fluidizing sand
Combustion air
Fluidized bed combustion rate
Incinerator volumetric liberation rate
Freeboard height
Peripheral equipment

PRINCIPLES OF FLUIDIZATION

In fluidized beds, the upflow gas suspends the solid particles. When solid particles are fluidized, two of the important factors are minimum fluidization velocity (Umf) and terminal velocity (Ut). Umf and Ut are decided by the characteristics of both the solid particles and the fluidizing gas. The density, shape, size and the space ratio of particles, and the density and viscosity (which is affected by temperature and pressure) of the gas decide the values of Umf and Ut.

When particles are fed into the vessel and the gas is moved in an upflow, if the velocity (Uo) of the gas is less than the minimum fluidization velocity, the particle bed will remain in a fixed state. When Uo is equal to or greater than Umf, the particles will be in the fluidizing state. If Uo is increased so that it becomes greater than terminal velocity, some of the particles which become separated from the fluidized bed surface are carried over the system with the gas.

REFRACTORY CONSTRUCTION

There are incinerators with a refractory construction that is a combination of shaped refractory and unshaped refractory, and incinerators of unshaped refractory only. The former is the more popular type of refractory construction. The following is an outline of refractory construction using a combination of shaped and unshaped refractory.

Because fluidizing sand creates the fluidized bed, the innermost wall of this section is constructed of insulating and wear-resistant firebrick to reduce wear. Behind this wall, there is a wall of insulating brick or castable insulating material, and a third wall made of insulating board.

Because the freeboard section is the combustion chamber and reaches temperatures between approximately 700 to 950°C, a refractory with superior antispalling and insulating characteristics must be selected.

FLUIDIZING SAND

The temperature of fluidizing sand should be maintained at about 750°C in the incinerator, but sand is cooled when discharged with incombustible materials from the incinerator bottom.

COMBUSTION AIR

Excess air ratio is the range of 1.5 to 2.0. Approximately 80 to 90% of the theoretical air quantity is supplied through air nozzles as the primary fluidizing air. The remainder is supplied to the freeboard section as secondary air.

Although the fluidizing air velocity must be a value between Umf and Ut, good fluidization for reactions and other factors is usually attained with Uo/Ut=0.2 to 0.6. If this ratio is higher than 0.6, disturbance in the bed will occur. If Uo/Ut=0.1 or less, then the fluidization will not be stable. The quantity of fluidizing air used per incinerator bed area is usually about 700 to 1,500 Nm^3/m^2h.

FLUIDIZED BED COMBUSTION RATE (Fb)

Fluidized bed combustion rate (Fb)

$$= \frac{\text{Combustion quantity (kg/h)}}{\text{Incinerator bed area } (m^2)}$$

Although the fluidized bed combustion rate depends on the quality of the refuse, it is usually between 400 to 600 kg/m^2h.

INCINERATOR VOLUMETRIC LIBERATION RATE (Lr)

Incinerator volumetric liberation rate (Lr)

$$= \frac{\text{Heat generation (kcal/h)}}{\text{Incinerator volume } (m^3)}$$

The incinerator volumetric liberation rate is within 7 to 10 x 10^4 $kcal/m^3h$.

FREEBOARD HEIGHT (Hf)

The guide to deciding freeboard height is the transport disengaging height (TDH). In the case of refuse incinerators, however, combustion in the freeboard and gaseous reaction of secondary air is important, so that sufficient reaction time must be allowed. Freeboard height is usually higher than splash zone height. About 4 - 7 seconds of resident time of gas in the bed and freeboard is sufficient.

PERIPHERAL EQUIPMENT

Refuse Shredder

Conventionally, it was considered better to shred refuse fed into fluidized bed incinerators. Recently, however, it has been found that efficient combustion is achievable without using a refuse shredder, so they are gradually going out of use in Japan.

Refuse Feeder

Refuse feeders must meet the following requirements:

(1) Must have a good seal to isolate the incinerator from the atmosphere.
(2) Must have a feed control of refuse.
(3) Must feed refuse uniformly and continuously.

As refuse feeders, rotary valves, rotary valves plus spreaders, single shaft screw conveyors and twin shaft screw conveyors are used.

Auxiliary Burner

Auxiliary burners are used to raise the temperature of fluidizing sand when the incinerator is started up, and to maintain combustion when the quality of refuse is low.

Startup burners must be located as close as the surface of fluidized bed to efficiently heat the fluidizing sand. However, wear caused by fluidizing sand must also be considered.

In many cases, oil guns which spray oil into the fluidized bed with compressed air to assist combustion of low quality refuse are installed.

Incombustibles Discharger

On the average, the amount of incombustibles in refuse in Japan is 4%. Incombustibles are discharged either continuously or intermittently with sand from the bottom of the incinerator. A vibrating screen installed below incombustible discharger, is used to seperate the incombustibles from the sand. Sand that has been separated is recycled. The amount of sand discharged with incombustibles is about four times the amount of the incombustibles.

OPERATING RESULTS

Operating results of a 40 ton/16 hour MSW fluidized bed incineration plant are presented. The plant is operated under the conditions shown in Figure 3 and Table 1.

PLANT STARTUP AND SHUTDOWN

The plant is operated semi-continuously, working for 16 hours operation and 8 hours shutdown.

When the fluidized bed temperature is 560 to 600°C, no auxiliary fuel is required for startup heating; the plant can be restarted simply charging waste. When the bed temperature is below 500°C, a bed burner must be used for the startup. When the bed temperature is between 500 and 560°C however, the incinerator can be heated to operating temperature by starting the bed burner and by charging waste at the same time, thus reducing oil consumption and startup time.

Normally, startup takes about 15 minutes, and the time required to attain normal operating temperature of 700°C or higher is not more than 30 minutes. The total time required for plant shutdown is about 30 minutes.

TEMPERATURE CONTROL

The temperature of the fluidized bed is held at 650°C and over, chiefly by regulating the waste feed rate. If the fluidized bed temperature falls below 600°C under a rated load, the bed burner is fired.

IGNITION LOSS

The ignition loss of ash sampled from the gas cooling tower and the electrostatic precipitator during plant operation is less than 1%. Ignition loss was measured by roasting the ashes at 600° ±25°C for 3 hours.

EMISSION ANALYSIS

Under the rated load, the emission characteristics met the emission standards satisfactorily and were as follows:

Dust concentration	0.032 g/Nm^3 (Dry gas)
Sulfur oxides (SOx)	14.9 ppm
Nitrogen oxides (NOx)	106 ppm
Hydrogen chloride (HCl)	178 ppm

FORMATION AND CONTROL OF DIOXIN

FORMATION OF DIOXIN

The amount of chlorinated compounds contained in gas and ash at the gas cooler outlet and the electrostatic precipitator (EP) outlet in the MSW fluidized bed incineration plant has been measured in an actual plant having capacity of 120 tons/day (2 x 60 tons/day). Table 2 gives the measurement result.

The measurement results show that organic chlorinated compounds such as PCDD, PCDF, chlorinated benzene, and lower chlorinated compounds are mostly formed in the EP, not in the incinerator.

Burning of refuse is a process in which high-molecular solids are pyrolytically decomposed into low-molecular compounds. At the normal operating temperature of the incinerator, about 700°C, low-molecular compounds are instantaneously oxidized; even chlorinated compounds formed by chlorination with chlorine contained in plastics easily decomposed and oxidized.

On the other hand, the operating temperature of the electrostatic precipitator is about 300°C. This temperature is too low for oxidization but adequate for chlorination of low-molecular compounds. The slow gas speed in the EP also helps the chlorination reaction.

Another question arises of which compounds would be chlorinated in the EP. The answer may be high-molecular organic compounds which remain unburnt in ash.

DUST COLLECTION AT LOW TEMPERATURE

The flue gas temperature at the incinerator outlet is about 750°C to 950°C. Flue gas is cooled down to 330°C rapidly by the gas cooling tower, and ash is collected at 300°C by the EP in normal operation.

In our experiment to collect dust at low temperature, the gas which comes out of the outlet of the gas cooling tower is sucked and collected by a vacuum pump on a filter made of glass felt which is fixed on a stainless steel holder (inner diameter: 200 mm). A collector tube is inserted behind the filter to measure the gas concentration after dust is collected. Gas of about 300°C is absorbed, but the temperature at which dust is collected was around 150°C because heat is dissipated through the stainless steel holder. The EP outlet gas and EP ash were also sampled for comparison.

Table 3 shows a comparison of the chlorinated compound concentrations after passing through a bag filter and an EP.

In the case of flue gas, all chlorinated compounds except polycyclic aromatic hydrocarbons are remarkably reduced by passing through the bag filter, when compared with the EP. Especially, 6, 7, and 8 CDFs concentration at the bag filter outlet are 1/50 to 1/100 of the concentration at the EP outlet.

As for the ash collected by the filter, dibenzofuran and 1 CDF showed higher values than EP ash. As the number of chlorine molecules in PCDDs and PCDFs increases, the filter to EP ratio becomes smaller. In other words, use of a filter is more effective to remove dioxin.

DECOMPOSITION OF CHLORINATED COMPOUNDS IN EP ASH

As mentioned above, dioxin still exists in ash collected by bag filters.

Laboratory scale experiments were conducted on the decomposition of dioxin. The result is shown in Table 4.

While the concentration of polycyclic aromatic hydrocarbon such as fluoranthene remained at the same level at 500°C or higher, chlorinated compounds were not detected at 500°C.

The amount of DD and DF which are no longer chlorinated compounds increased markedly at 400°C but the total amount of 0 to 8 compounds decreased. Also, the amount of DD decreased at 500°C, and it was no longer detected at 600°C. The amount of DF increased at 500°C, and no change was observed at 600°C.

CONTROL OF DIOXIN

Since PCDD, PCDF, and chlorinated benzene are scarcely formed in the incinerator, the flue gas treatment temperature after incineration is important to prevent generation of chlorinated compounds.

If chlorinated compound concentrations are high after incineration, the combustion condition of the incinerator must be checked.

If ash is collected at low temperature, the volatization of PCDD and PCDF, which are adhered to ash, will decrease and the dioxin concentration of the flue gas can be decreased.

Chlorinated compounds remaining in ash can be decreased to a negligible level by high temperature treatment.

CONCLUSION

According to our experiments, the procedures for reduction of the amount of organic chlorinated compounds which are formed in the incineration facilities are to burn MSW completely in the incinerator, to cool flue gas rapidly, and to collect ash at low temperature. Chlorinated compounds in ash can be decomposed and removed at a high temperature.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

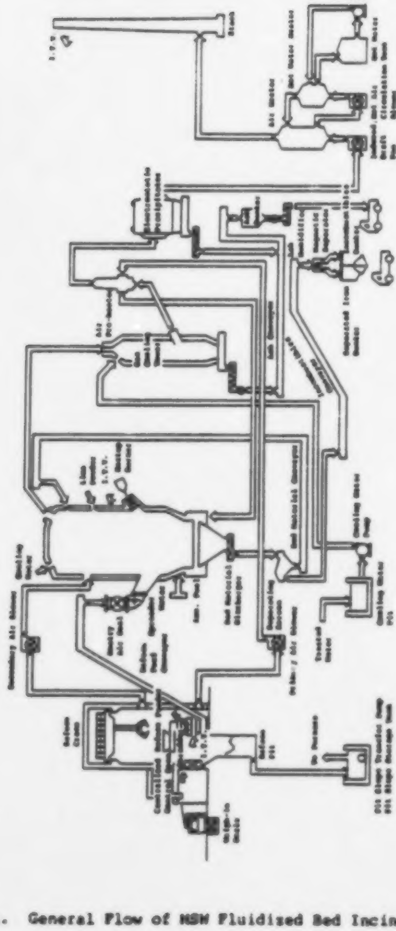

Figure 1. General Flow of MSW Fluidized Bed Incineration Plant

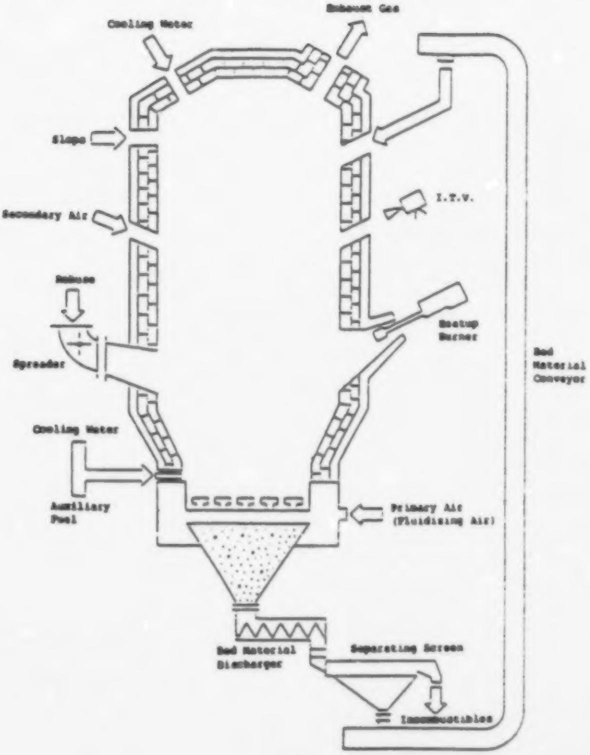
Exhaust Gas
Cooling Water
Slope
I.T.V.
Secondary Air
Refuse
Heatup Burner
Spreader
Bed Material Conveyor
Cooling Water
Auxiliary Fuel
Primary Air (Fluidizing Air)
Bed Material Discharger
Separating Screen
Incombustibles

Figure 2. MSW Fluidized Bed Incinerator

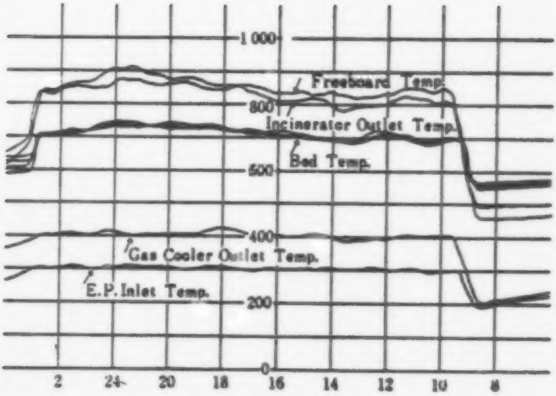

Figure 3. Temperature Chart of MSW Fluidized Bed Incinerator

Table 1. Operating Date

		Test 1	Test 2	Test 3
Feed rate	kg/h	2,510	2,730	2,550
Calorific value	kJ/kg	7,200	5,020	7,370
Bed load	kg/m^2,h	423	460	430
Bed temp.	°C	700	680	730
Incinerator outlet temp.	°C	850	760	880
Primary air temp.	°C	200	250	180
Auxiliary fuel	ℓ/h	0	0	0
Fluidizing air	Nm^3/h	4,700	4,700	4,700
Secondary air	Nm^3/h	3,100	2,300	3,200

Table 2. Chlorinated Compound Concentration in Flue Gas and in Ash

	Flue gas ($\mu g/m^3$)			Ash (ng/g)		
	EP inlet	EP outlet	Measure limit	Gas cooler	EP	Measure limit
o PCDD						
4CDD	nd	nd	nd<0.1	nd	nd	nd<0.5
5CDD	nd	nd	nd<0.5	nd	nd	nd<1.0
6CDD	nd	0.46	nd<0.1	nd	17	nd<0.5
7CDD	nd	1.4	nd<0.2	nd	96	nd<1.0
8CDD	nd	1.7	nd<0.2	nd	150	nd<1.0
o PCDF						
4CDF	nd	2.1	nd<0.1	nd	26	nd<1.0
5CDF	nd	0.76	nd<0.1	nd	110	nd<5.0
6CDF	nd	13	nd<0.5	nd	320	nd<2.0
7CDF	nd	6.2	nd<0.5	nd	430	nd<2.0
8CDF	nd	6.2	nd<0.5	nd	440	nd<2.0
o Chlorobenzene						
Trichlorobenzene	1.2	8.2		2.1	12	
Tetrachlorobenzene	2.4	28		5.1	37	
Pentachlorobenzene	5.0	150		15	250	
Hexachlorobenzene	1.3	83		12	200	
o Aromatics						
Anthracene, phenantrene	54	46		160	57	
Fluoranthene	36	31		123	15	
Pyrene	35	9.1		123	6.0	

Table 3. Comparison of Chlorinated Compound Concentration After Passing Through a Bag Filter and an EP

	Flue gas ($\mu g/m^3$)			Ash (ng/g)		
	EP outlet	Filter outlet	Filter/ EP	EP	Filter	Filter/ EP
o PCDD						
1CDD	nd<0.1	nd<0.1	-	nd<0.2	nd<0.8	-
2CDD	nd<0.05	nd<0.05	-	nd<0.1	nd<0.4	-
3CDD	nd<0.05	nd<0.05	-	nd<0.1	nd<0.4	-
4CDD	nd<0.1	nd<0.1	-	35	nd<0.8	<0.02
5CDD	nd<0.1	nd<0.2	-	109	nd<1.0	<0.01
6CDD	2.7	nd<0.1	<0.04	140	nd<0.8	<0.005
7CDD	6.8	nd<0.2	<0.03	372	nd<1.0	<0.004
8CDD	5.1	nd<0.2	<0.04	195	nd<1.0	<0.005
o PCDF						
1CDF	13	13	1.0	24	44	1.83
2CDF	4.2	3.5	0.83	84	46	0.55
3CDF	5.2	nd<0.2	<0.04	151	25	0.17
4CDF	23	nd<0.2	<0.01	577	86	0.15
5CDF	58	nd<0.2	<0.003	1350	63	0.05
6CDF	161	1.6	0.01	3160	21	0.01
7CDF	145	2.9	0.02	3130	43	0.01
8CDF	34	nd<0.2	<0.006	744	nd<1.0	<0.001
o Chlorobenzene						
Trichlorobenzene	179	29	0.11	157	48	0.31
Tetrachlorobenzene	359	16	0.04	383	79	0.21
Pentachlorobenzene	399	6.7	0.02	639	110	0.17
Hexachlorobenzene	89	1.0	0.01	240	nd<1.0	<0.004
o Aromatics						
Anthracene, phenantrene	784	792	1.01	2080	559	0.27
Fluoranthene	49	37	0.76	930	291	0.31
Pyrene	6.8	10	1.47	392	87	0.22
Dibenzofuran	226	203	0.90	92	263	2.88

Table 4. Relation between Heat Treatment Temperature and Chlorirated Compound Concentration in Ash

ng/g

	Non-treatment	100°C	200°C	300°C	400°C	500°C	600°C
o PCDD							
DD	nd<0.5	nd<0.5	nd<0.5	nd<0.5	45	4.5	nd<0.5
1CDD	nd<0.2	nd<0.2	nd<0.2	4.1	54	nd<0.2	nd<0.2
2CDD	nd<0.2	nd<0.2	nd<0.2	1.9	6.5	nd<0.2	nd<0.2
3CDD	nd<0.2	nd<0.2	nd<0.2	5.7	2.3	nd<0.2	nd<0.2
4CDD	7.2	9.2	0.4	102	5.8	nd<0.2	nd<0.2
5CDD	1.0	nd<0.2	nd<0.2	52	nd<0.2	nd<0.2	nd<0.2
6CDD	21	23	17	44	0.8	nd<0.2	nd<0.2
7CDD	120	133	103	32	nd<0.2	nd<0.2	nd<0.2
8CDD	232	258	214	46	nd<0.2	nd<0.2	nd<0.2
o PCDF							
DF	nd<0.5	nd<0.5	nd<0.5	4.5	133	331	372
1CDF	nd<0.5	nd<0.5	nd<0.5	12	162	nd<0.5	nd<0.5
2CDF	nd<0.5	nd<0.5	nd<0.5	71	230	nd<0.5	nd<0.5
3CDF	nd<0.5	nd<0.5	nd<0.5	154	72	nd<0.5	nd<0.5
4CDF	21	29	19	892	47	nd<0.5	nd<0.5
5CDF	125	154	11	600	20	nd<0.5	nd<0.5
6CDF	337	408	295	291	19	nd<0.5	nd<0.5
7CDF	1460	1600	1260	280	31	nd<0.5	nd<0.5
8CDF	1160	1350	1090	140	10	nd<0.5	nd<0.5
o Chlorobenzene							
Trichlorobenzene	28	17	17	240	619	nd<0.2	nd<0.2
Tetrachlorobenzene	78	59	67	1120	38	nd<0.2	nd<0.2
Pentachlorobenzene	192	143	169	185	12	nd<0.2	nd<0.2
Hexachlorobenzene	187	161	151	55	2.9	nd<0.2	nd<0.2
o Aromatics							
Anthracene, phenantrene	260	240	183	122	150	360	460
Fluoranthene	129	140	136	91	67	98	138
Pyrene	68	70	55	39	64	107	121

DEVELOPMENT OF COMBUSTION GUIDELINES TO MINIMIZE AIR EMISSIONS FROM MUNICIPAL WASTE COMBUSTORS

EPA's Office of Research and Development (ORD) is currently supporting the Office of Air Quality Planning and Standards (OAQPS) in the development of air emission standards for new and existing municipal waste combustors (MWCs). One of the key inputs from ORD is the development of combustion guidelines to ensure that emissions of trace organics and other pollutants are minimized. This paper will discuss the background and technical basis for the combustion guidelines that are developed for each combustor technology. In addition, discussion related to design and operating features of the various classes of combustors that comprise the existing and projected populations will be included, along with an analysis of how combustor technology dictates the application of the guidelines. Performance data will be presented in the context of providing support for the elements of the guidelines, and the effects of the guidelines on potential emission levels will be discussed.

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SOURCE TESTING OF BIOMASS CO-GENERATION FACILITIES

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ABSTRACT

Over the last six years staff of the Compliance Division of the California Air Resources Board have tested numerous biomass co-generation facilities in California primarily for particulate matter emissions but also for emissions of nitrogen oxides and carbon monoxide. This paper describes the specific sources tested, the method used by the sources for controlling particulate matter or other emissions (as necessary), the source test methodology, the results of the tests, and other pertinent information. Findings from the tests conducted suggest certain types of control devices and combustion systems may be more effective than others in controlling emissions from biomass co-generation facilities.

INTRODUCTION

The Air Resources Board (ARB) was created by the California Legislature for the purpose of controlling air pollution and enhancing air quality through the State. It works closely with the Environmental Protection Agency (EPA) and also with local air pollution control districts (APCDs) within the State which have primary authority over the control of air pollutant emissions from stationary sources within their specific jurisdiction. The ARB is given authority to review local districts to ensure proper implementation of the districts' control programs and to provide technical assistance. The ARB is given primary statutory authority to establish and to enforce standards that govern emissions from motor vehicles.

The ARB Compliance Division provides assistance to the local districts in enforcing the various air pollutant emission standards applied to stationary sources. This responsibility leads to the testing of many stationary sources located within a local district to ensure compliance. As a result of financial limitations, many local districts do not have the resources or expertise to conduct compliance source tests and, consequently, the ARB or an independent source testing company will be requested to conduct the testing.

The scope of this paper is limited to the data compiled from tests of biomass co-generation facilities performed by the ARB Compliance Division since 1982. Its purpose is to describe the operations and characteristics of the sources tested, the type and amount of pollutants measured, and assess which of the sources achieved the best results.

SOURCES TESTED

Nineteen biomass co-generation sources were tested by the ARB Compliance Division. Some of these were tested on more than one occasion to determine both gaseous and particulate matter emissions. The sources operating parameters, including steam production, type of biomass fuel used, fuel moisture content, the firing rate, combustion systems and control equipment are specifically described for each source. These sources all share similar operations and characteristics. These similarities are discussed first.

GENERAL DESCRIPTION OF SOURCE OPERATION AND CHARACTERISTICS

All sources tested were biomass co-generation facilities. These are sources which utilize the heat of combustion to generate steam and in turn may use the steam to generate electricity, i.e. co-generation. They are termed biomass because they use as their fuel supply agricultural waste products, forest waste products, or similar organic wastes. Biomass fuel includes agricultural cuttings, nut shells and hulls, fruit pits, and the wood waste associated with forest harvesting such as bark, sawdust, milling waste, and untreated construction wood products.

The facilities tested used all or any combination of these types of biomass fuels and may also supplement biomass fuels with fuel oil or natural gas to enhance combustibility or counteract the effects of high moisture content fuel in the combustion process. (The determination as to auxiliary fuels which could/can be used falls within the local APCD's jurisdiction). Regardless of the type of biomass fuel used, all fuel was broken down into small chips or slivers varying in size up to 1.5 inches prior to its use to facilitate handling, usually by conveyor belt or screw conveyor, and enhance fuel combustion. Some sources may also have subjected the fuel to a drying process in order to lower moisture content prior to combustion.

The types of biomass fuel used, to a certain degree, depended on the source's location. For instance, facilities located in the coastal or mountain regions utilize more "hog" fuel (sawmill and forest residue consisting of sawdust, bark, chips, shavings, and the mills' untreated wood products such as waste lumber) from lumber mills than do those facilities located in the valley regions of the State, which burn more commercial and agricultural related biomass fuels such as orchard trimmings, nut shells, and fruit pits.

All sources tested use boilers to transfer the heat from the combustion process to water in order to generate steam. All boilers consist essentially of a burner, firebox, heat exchanger, and a means of creating or directing a flow of gases through the unit. Steam is produced by the vaporization of water passing through cells or tubes that have been heated from the combustion of fuel. The burner is the device used to ignite the fuel. For the majority of sources tested this was a grate upon which fuel was continuously dropped and combusted. The firebox is the furnace where the burner is located and hence where combustion takes place. The cells or tubes must pass through the firebox in order to be heated. The heat exchanger is the collection of cells or tubes themselves, transferring heat from the combustion exhaust to the water to create steam.

The source of pollutants in a steam generation system is, of course, the combustion process. In turn, various types of equipment, processes and operating parameters can be used to enhance the combustion process and reduce the amount and types of pollutants generated.

The type of biomass fuel burned plays an important role in the combustion process, prescribing the amount needed to maintain firebox temperatures. Similarly, the fuel's moisture content affects the combustion process, using energy otherwise available to heat the firebox to vaporize the water in the fuel (thermal loss). To mitigate this effect some sources now dry their fuel prior to its use by subjecting it to a drying process or mixing it with fuel of a lower moisture content. The size of the fuel may also affect the ease and rate at which fuel is combusted (i.e. the average size of the woodwaste chip, sawdust, pits, nut shells, etc.). Fuel that is of a smaller effective diameter may have a short residence time (length of time fuel remains on the grate burner) and, consequently, may leave the grate burner before being completely burned by being forced away from the grate by underfire air. Fuel that is of a larger size may have a residence time that proportionately exceeds the amount of heat the fuel can produce, remaining on the grate burner as a mass of ash or carbon cinder and occupying space otherwise available to fresh unburned fuel.

In order to ensure that the fuel is completely burned and releasing the maximum amount of heat, the burning fuel is stoked. This may be accomplished by several processes or equipment. The combustion air may be forced up through the burning fuel or directed at or above the burning fuel from the sides of the firebox to enhance the intimate contact between fuel and the oxygen needed for combustion to take place. The equipment on which the fuel is burned may remain in motion, such as a travelling grate, and hence provide better oxygen/fuel contact, as well as aiding in the removal of fuel that has exceeded the ideal residence time.

With regard to oxygen, it is necessary to provide sufficient oxygen (referred to as excess air) to the firebox to ensure complete combustion of the fuel. However, any air above the theoretical amount needed for combustion represents a thermal loss since the heat otherwise available to heat the firebox is being used to heat excess air. For this reason the excess air made available to the the firebox is regulated. (Usually by regulating the amount air that is forced up through the fuel.) In addition to thermal loss, the presence of excess air enhances the formation of gaseous pollutants such as oxides of nitrogen and carbon monoxide, which are inherent in the combustion process. Excess air is but one parameter affecting the creation of such pollutants.

With few exceptions, all boiler systems tested employed a combustion process whereby fuel was continuously dropped upon a grate located in th firebox. The majority of these utilized a stationary grate; however, some employed a moving grate to stoke the fire and enhance combustibility. Combustion air was forced up from beneath the grate and may have been preheated (underfire air). Some boiler systems may continuously separate the carbon cinder from the ash of burned fuel and reinject the carbon-cinder portion (along with unburned fuel and the cinder portion of material caught in the particulate matter control devices) above the grate (reinjection air) to enhance combustion. Air may also be injected above the grate to enhance or stage combustion (overfire air). The majority of boiler systems utilized multiple-cyclone separators (multiclone collectors) in combination with wet scrubbers, baghouses, or electrostatic precipitators (ESPs) to control particulate matter in the exhaust.

SPECIFIC DESCRIPTION OF SOURCE OPERATIONS AND CHARACTERISTICS

Source #1 operates a steam boiler with a steam production capacity of 110,000 lbs/hr. The boiler is typically fired on hog fuel with a moisture content of approximately 50%. The combustion system utilizes a stationary grate with preheated underfire and overfire air. The system also employs reinjection air. Particulate matter emissions are controlled with multiclone collection in combination with an ESP.

Source #2 operates a steam boiler with a steam capacity of 100,000 lbs/hr. The boiler is typically fired on hog fuel with a moisture content of approximately 50%. The combustion system utilizes a stationary grate with preheated underfire and overfire air. The system also employs reinjection air. Particulate matter emissions are controlled with multiclone collection in combination with an ESP.

Source #3 operates a steam boiler with a rated steam capacity of 110,000 lbs/hr. The boiler is typically fired on hog fuel with a moisture content of approximately 50%. The combustion system utilizes a travelling grate with preheated underfire and overfire air. The system also employs reinjection air. Particulate matter emissions are controlled with multiclone collection and an ESP.

Source #4 operates a steam boiler with a rated steam capacity of 150,000 lbs/hr. The boiler is typically fired on hog fuel with a moisture content of approximately 54%. The combustion system utilizes a travelling grate with preheated underfire and overfire air. Particulate matter emissions are controlled with multiclone collection and an ESP.

Source #5 operates a steam boiler with a rated steam capacity of 100,000 lbs/hr. The boiler is typically fired on a combination of hog fuel, construction wood waste, almond and walnut shells, peach and olive pits, and vineyard and orchard prunings with an overall moisture content of approximately 20 to 25%. The combustion system utilizes a stationary grate with preheated underfire and overfire air. The system also employs reinjection air. Particulate matter emissions are controlled with multiclone collection and an ESP.

Source #6 operates a steam boiler with a rated steam capacity of 125,000 lbs/hr. The boiler is typically fired on a combination of hog fuel, construction wood waste, almond and walnut shells, peach and olive pits, and vineyard and orchard prunings with an overall moisture content of approximately 42%. The combustion system utilizes a travelling grate with preheated under and overfire air. The system also employs reinjection air. Particulate matter emissions are controlled with multiclone collection and an ESP.

Source #7 operates a steam boiler with a rated steam capacity of 210,000 lbs/hr. The boiler is typically fired on hog fuel and on sawdust separately with approximate moisture contents of 53% and 20%, respectively. The combustion system utilizes a stationary grate to burn the hog fuel with preheated underfire and overfire air. The sawdust fuel is injected directly above the grate with the reinjection air. Particulate matter emissions are controlled with multiclone collection and a wet scrubber.

Source #8 operates a steam boiler with a rated steam capacity of 105,000 lbs/hr. The boiler is typically fired on hog fuel with a moisture content of approximately 40% The combustion system utilizes a stationary grate with preheated underfire and overfire air. The system also employs reinjection air. Particulate matter emissions are controlled with multiclone collection and a wet scrubber.

Source #9 operates a steam boiler with a rated steam capacity of 105,000 lbs/hr. The boiler is typically fired on hog fuel with a moisture content of approximately 40%. The combustion system utilizes a stationary grate with preheated underfire and overfire air. The system also employs reinjection air. Particulate matter emissions are controlled with multiclone collection and a wet scrubber.

Source #10 operates a steam boiler with a rated steam capacity of 140,000 lbs/hr. The boiler is typically fired on hog fuel with a moisture content of approximately 40%. The combustion system utilizes a travelling grate with preheated underfire and overfire air. The system also employs reinjection air. Particulate matter emissions are controlled with multiclone collection and a wet scrubber.

Source #11 operates a boiler with a rated steam capacity of 160,000 lbs/hr. The boiler is typically fired on hog fuel with a moisture content of approximately 48%. The combustion system utilizes a stationary grate with preheated under and overfire air. The system also employs reinjection air. Particulate matter emissions are controlled by multiclone collection and a wet scrubber.

Source #12 operates a boiler with a rated steam capacity 15,000 lbs/hr. The boiler is typically fired on hog fuel with a moisture content of 10 to 20%. The combustion system utilizes a stationary grate with preheated underfire and overfire air. Particulate matter emissions are controlled with multiclone collection.

Source #13 operates a boiler with a rated steam capacity of 34,000 lbs/hr. The boiler is typically fired on specially ground hog fuel that varies in size from 1/64 to 1/4 of an inch with a moisture content of approximately 10%. The combustion system differs substantially from other sources tested in that fuel is combusted more like a liquid or a gas fuel would be. The hog fuel is continuously mixed with non-heated underfire air via injection onto a dispersion plate and ignited. Particulate matter emissions are controlled with multiclone collection and a wet scrubber.

Source #14 operates a boiler with a rated steam capacity of 125,000 lbs/hr. The boiler is typically fired on a combination of almond shells, hog fuel, macerated tree trimmings, and pits with an average moisture content of 12%. The combustion system utilizes a stationary grate with preheated underfire and overfire air. The system also employs reinjection air. Particulate matter emissions are controlled with multiclone collection and a baghouse.

Source #15 operates a boiler with a rated steam capacity of 125,000 lbs/hr. The boiler was fired on a combination of almond shells, hog fuel, macerated tree trimmings, and pits with an average moisture content of 12% and at an approximate rate of 20,800 lbs/hr. The combustion system utilizes a stationary grate with preheated underfire and overfire air. The system also employs reinjection air. Particulate matter emissions are controlled with multiclone collection and a wet scrubber.

Source #16 operates a boiler with a rated steam capacity of 60,000 lbs/hr. The boiler was fired on hog fuel with a moisture content of approximately 43%. The combustion system utilizes a stationary grate with preheated underfire and overfire air. Particulate matter emissions are controlled with multiclone collection.

Source #17 operates a boiler with a rated steam capacity of 40,000 lbs/hr. The boiler is typically fired on hog fuel with a moisture content of approximately 35%. The combustion system utilizes a stationary grate with preheated underfire and overfire air. Particulate matter emissions are controlled with multiclone collection.

Source #18 operates a boiler with a rated steam production of 12,000 lbs/hr. The boiler is typically fired on olive pits with a an average moisture content of 55%. The combustion system differs substantially from other sources tested in that it does not utilize a grate, but instead uses a bed of hot sand (often referred to as a fluidized bed). Preheated underfire air is injected up through the bed of sand, thereby causing it to remain in motion, hence "fluid." The fuel is continuously dropped upon the bed of sand, where it is ignited. Preheated overfire air is also employed. Particulate matter emissions are controlled with a baghouse.

Source #19 operates a boiler with a rated steam capacity of 50,000 lbs/hr. The boiler is typically fired on a combination of hog fuel, tree trimmings, and peach and olive pits with an average moisture content of 11%. The combustion system utilizes a stationary grate with preheated underfire and overfire air. Particulate matter emissions are controlled with multiclone collection and a baghouse.

SOURCE TEST METHODOLOGY

Of the 29 tests conducted by the Compliance Division 20 measured particulate matter emissions, and the remaining tests measured gaseous emissions. Testing of all sources followed the same procedures.

PARTICULATE MATTER EMISSIONS

Particulate matter emissions were evaluated using ARB Test Method #5, Particulate Matter Emissions. This method is comparable to EPA Reference Method #5. Essentially, particulate matter is withdrawn isokinetically from the stack and collected on a glass fiber filter maintained at a temperature of approximately 248 degrees Fahrenheit (120 degrees Celsius). The particulate mass is then determined gravimetrically after removal of uncombined water.

All condensable particulate matter that passes through the glass filter at the filtration temperature is cooled and caught in an impinger train. All filterable and condensable particulate matter as well as material rinsed from the sampling nozzle, probe, filter holder, and glass connections between the filter holder and the third impinger is defined by the ARB as particulate matter. This definition is subject to change within local APCDs. However, all biomass facilities tested by the ARB were located in districts accepting ARB's definition.

As noted above, ARB Test Method #5 and EPA Reference Method #5 are essentially the same, differing only in the definition of particulate matter. For purposes of determining compliance, EPA Method #5 defines particulate matter (mass) as any material that condenses at the filtration temperature. This definition therefore excludes any condensable particulate matter passing through the filter, i.e. the "back-half" catch. By comparison, ARB's definition leads to a more restrictive result by including particulate matter otherwise not included under EPA Method #5.

GASEOUS EMISSIONS

Gaseous emissions, including oxides of nitrogen, carbon monoxide, carbon dioxide, oxides of sulfur, total hydrocarbon, and oxygen, were evaluated using ARB Test Method #100, Continuous Gaseous Emissions Stack Sampling. Essentially, a sample of the boiler's exhaust was extracted, conditioned, and analyzed continuously by instruments. The measurements made by the continuous analyzers were used to determine average emissions concentrations. A nondispersive infrared continuous analyzer was used to measure carbon dioxide and carbon monoxide. A chemiluminescent analyzer was used to measure oxides of nitrogen. Oxygen was measured using a paramagnetic analyzer, and flame ionization detection and ultraviolet absorption were used to measure total hydrocarbon and sulfur dioxide, respectively.

TEST RESULTS

PARTICULATE MATTER EMISSIONS

20 tests to determine the amount of particulate matter in the exhaust were conducted on 15 sources. The test results are listed below in Table 1 along with other pertinent information about the sources. Sources that were tested on more than one occasion have respective results listed in sequence. The emission results reported are the test average corrected to 12% carbon dioxide. Also listed, adjacent to emission results, are applicable emission limitations given in corresponding units. The boiler's steam production is also given as an average over the testing period.

Under the "Combustion System" column of the following table, note that all sources except #13 utilize preheated underfire and overfire air. Also, unless specified as travelling or other, a source is deemed to have a stationary grate in addition to that listed.

TABLE 1. PARTICULATE MATTER EMISSIONS RESULTING FROM SOURCE TESTING VARIOUS BIOMASS CO-GENERATION FACILITIES

SOURCE	STEAM PRODUCTION (lbs/hr)	FUEL		COMBUSTION SYSTEM	CONTROL DEVICE	PARTICULATE MATTER		LIMIT
		TYPE	% H2O			grain/sdcfm	lb/hr	
#5	84,000	hog	25	reinject air	multi + ESP	0.008	2.65	0.016 4.72
#6	117,000	85% hog 15% ag	42	travelling grate + reinject air	multi + ESP	0.023	9.33	0.016 6.41
#6	120,000	85% hog 15% ag	42	same	same	0.057	24.2	0.016 6.41
#7	200,000	hog sawdust	53 20	reinject air + sawdust	multi + wetscrb	0.046	99.4	0.10 none
#8	48,000	hog	40	reinject air	multi + wetscrb	0.030	10.0	0.200 none
#9	59,000	hog	40	reinject air	multi + wetscrb	0.043	16.1	0.200 none
#10	86,000	hog	40	travelling grate + reinject air	multi + wetscrb	0.022	9.1	0.10 none
#11	135,000	hog	48	reinject air	multi + wetscrb	0.084	23.6	0.10 none
#12	1,000	hog	15		multi	0.11	2.2	0.20 10.0
#13	22,000	fine hog	10	liquid fuel like combustion	multi + wetscrb	0.21	11.6	0.10 none
#13	22,000	fine hog	10	same	same	0.15	9.6	0.10 none
#14	125,000	70%almond shell 30% ag	12		multi + baghous	0.072	34.0	0.02 none
#14	121,000	30%almond 60% hog 10% pits	12		same	0.086	---	0.02 none
#15	90,000	70%almond 30% hog	12		multi + wetscrb	0.067	---	0.018 none

TABLE 1. Continued

#15	97,000	almond shell	12		same	0.0476	---	0.018 none
#16	58,000	hog	43		multi	0.130	29.4	0.30 none
#17	33,000	hog	35		multi	0.097	12.9	0.30 none
#18	12,000	olive pits	55	fluidized bed	baghous	0.266	8.2	0.10 10.0
#19	43,400	90% peach pits 10% hog	11		multi + baghous	0.065	7.74	0.10 8.87
#19	27,700	same	11		same	0.032	2.90	0.10 6.80

Particulate matter emissions were lowest when multiclone collection was used in conjunction with an ESP. Source #5 and #6 both utilized such control devises. Source #5 was shown to emit 0.032 pounds of particulate matter per 1000 lbs of steam produced (lbs/k steam). The first test conducted on source #6 revealed it to be emitting 0.080 lbs/k steam. The second test conducted on source #6 revealed it to be emitting 0.202 lbs/k steam. Although the second test does not compare with the first, it can be distinguished. During the second test, the source operators alleged the control equipment was malfunctioning, resulting in the higher emissions.

Sources utilizing multiclone collection in conjunction with baghouses performed slightly better for controlling particulate matter emissions than those utilizing multiclone collection in conjunction with wet scrubbers. Sources using the former were found to be emitting as low as 0.105 lbs/k steam and as high as 0.272 lbs/k steam. Sources using the latter were found to be emitting as low as 0.106 lbs/k steam and as high as 0.527 lbs/k steam.

One source was tested that used only a baghouse. This source was found to be emitting 0.683 lbs/k steam. This number falls between the low and high for sources utilizing multiclone collection: 0.391 lbs/k steam and 2.2 lbs/k steam.

GASEOUS EMISSIONS

Gaseous emission were measured from 9 of the 19 sources tested. Although oxides of nitrogen (NOx), carbon monoxide (CO), carbon dioxide, oxides of sulfur, total hydrocarbon, and oxygen were measured, only oxides of nitrogen and carbon monoxide will be reported. Oxides of sulfur and total hydrocarbon were found to be negligible on the nine sources tested.

As with particulate matter emission results, gaseous emission results are presented in tabular form along with other pertinent source information and applicable emission limitations (see Table 2). Gaseous emissions results listed are given as test averages at stack conditions (not calculated at a given oxygen percentage). Steam production information is given as an average over the test period. Emission limitations are listed adjacent to the corresponding emission results and are given in lbs/hr.

With regard to the "combustion system" column of Table 2., all sources tested utilized preheated underfire and overfire air. Unless specified as utilizing a travelling grate, all sources utilized a stationary grate in addition to that listed.

TABLE 2. NOX AND CO EMISSIONS MEASURED BY SOURCE TESTING VARIOUS BIOMASS CO-GENERATION FACILITIES

SOURCE	STEAM PRODUCTION (lbs/hr)	FUEL		COMBUSTION SYSTEM	CONTROL DEVICE	GASEOUS EMISSIONS			LIMIT
		TYPE	% H2O				ppm	lb/hr	
#1	120,000	hog	50	reinject air	multi + ESP	NOx: CO :	69.5 851	27.86 207.7	none none
#2	180,000	hog	58	reinject air	multi + ESP	NOx: CO :	72 2248	41.44 787.5	none none
#3	118,000	hog	50	travelling grate + reinject air	multi + ESP	NOx: CO :	75 928	60.42 188.0	none none
#4	90,000	hog	54	travelling grate	multi + ESP	NOx: CO :	73 55.7	28.70 13.33	none none
#5	108,000	hog	25	reinject air	multi + ESP	NOx: CO :	70 103	19.5 17.5	18.5 59.9
#6	120,000	85% hog 15% ag	42	travelling grate + reinject air	multi + ESP	NOx: CO :	42 220	15.6 49.8	22.21 99.3
#12	1,000	hog	15		multi	NOx: CO :	285 ---	8.2 ---	140 ---
#14	118,600	70%almond shells 30% ag	12	reinject air	multi + baghous	NOx: CO :	148 ___	___ ___	none none
#15	100,000	almond shells	12	reinject air	multi + wetscrb	NOx: CO :	139 849	26.72 99.35	none none

Based upon the the above listed gaseous emission results, certain findings can be made about the combustion systems utilized by the sources. If the amount of pollutants, measured in pounds per hour, is compared with the amount of steam produced, measured in 1000 pounds per hour, the effectiveness of the combustion system can be evaluated. This comparison, however, does not take into account many parameters such as the age of the system equipment, its specific design, or whether the boiler was operating below or above its rated steam production capacity.

NOx results were lowest for source #6, emitting 0.130 pounds of NOx per 1000 pounds of steam produced (lbs/k steam). The combustion system for this source utilized a travelling grate with preheated underfire and overfire air, as well as reinjection air. The NOx results for sources #1, #2, #5, and #15 are consistent, varying from 0.181 lbs/k steam to 0.267 lbs/k steam. All of these sources used stationary grates with preheated underfire and overfire air and reinjection air. NOx emissions were highest for source #12, which emitted 8.2 lbs/k steam. Note, source #15 was operating at very low steam production capacity when tested.

CO results were lowest for source #4, emitting 0.148 lbs/k steam. CO emissions were highest for #2, emitting 4.375 lbs/k steam.

CONCLUSIONS

Significant conclusions about the effectiveness of a particular type of boiler combustion system or control devise would be inappropriate in light of the limited amount and specificity of tests conducted. However, based upon a comparison of the emissions and steam production, it was possible to make certain limited findings. These findings are based upon the amount of pollutant per quantity of steam produced without regard to many parameters such as the age of the system equipment, its specific design, or whether the boiler was operating below or above its rated steam production capacity. The comparison revealed that particulate matter emissions from biomass cogeneration facilities were controlled best by multiple cyclone separators used in conjunction with electrostatic precipitators. With regard to gaseous emissions, oxides of nitrogen were controlled best by a source utilizing a combustion system equipped with a travelling grate burner having preheated underfire and overfire air and having air and carbon cinders reinjected over the grate. Carbon monoxide emissions were controlled best by a source utilizing a combustion system equipped with a travelling grate burner having preheated underfire and overfire air, but without reinjected air. The type of biomass fuel used and its moisture content did not appear to have any significant effect on the resulting pollutants.

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DEVELOPMENT OF GOOD COMBUSTION PRACTICES TO MINIMIZE AIR EMISSIONS FROM MUNICIPAL WASTE COMBUSTORS

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ABSTRACT

The U.S. Environmental Protection Agency's (EPA's) Office of Research and Development (ORD) is currently supporting the EPA's Office of Air Quality Planning and Standards (OAQPS) in the development of air emission standards and guidelines for new and existing municipal waste combustors (MWCs). One of the key inputs from ORD is the development and recommendation of good combustion practices (GCP) to ensure that formation of trace organics and other pollutants is minimized and their destruction is maximized. GCP recommendations for controlling trace organics, published by EPA in June 1987, provided criteria for design, operation, and verification of mass burn waterwall, refuse-derived fuel fired, and modular starved air MWCs. This paper discusses current activities in revising the initial GCP recommendations and in developing GCP for other MWC technologies. Performance data are presented which support the use of GCP to control trace organic emissions.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for presentation and publication.

INTRODUCTION

On July 7, 1987, the U.S. Environmental Protection Agency (EPA) published an advance notice of proposed rulemaking (ANPRM) in the Federal Register (1). This notice announced EPA's intention to regulate air pollution emissions from municipal waste combustors (MWCs). The MWC emission standards and guidelines, which will address, respectively, new and existing facilities, are scheduled for proposal in November 1989, and promulgation in December 1990. EPA's decision to regulate MWC emissions was based, in part, on a comprehensive study of municipal waste combustion, which included evaluation of the health and environmental risks associated with MWCs and an assessment of the criteria and hazardous air pollutant emission reduction potential of both combustion process controls and flue gas cleaning (FGC) technologies (2-5). This study concluded that 1) new MWCs are likely to be significant sources of criteria pollutant emissions; 2) MWC emissions include a wide variety of constituents including chlorinated dibenzo-p-dioxins and dibenzofurans (CDD/CDF), other potentially toxic organics, heavy metals, and acid gases that can adversely affect public health and welfare; and 3) the MWC industry has the potential for significant growth over the next 20 years.

Concurrently with the ANPRM, EPA's Office of Air Quality Planning and Standards (OAQPS) issued guidance to EPA's Regional Offices for permitting new MWCs prior to promulgation of the revised New Source Performance Standard (NSPS) (6). This guidance specified that all new MWCs use good combustion practice (GCP) and the appropriate FGC technology to ensure adequate control of air pollution emissions. Appropriate FGC technology was defined as the use of a dry scrubber (spray dry adsorber) in combination with a fabric filter (FF) or electrostatic precipitator (ESP). Although the criteria for achieving good combustion were not specified, EPA's Regional Offices and state/local permitting agencies were referred to recommendations for good combustion provided in the report, Municipal Waste Combustion Study: Combustion Control of Organic Emissions (4). Subsequently, EPA's Office of Research and Development (ORD) was asked to review the GCP recommendations in the Report to Congress and to expand GCP to the full range of MWC technologies. In response, ORD initiated a study to investigate MWC design and operating conditions under which CDD/CDF are created and destroyed and develop GCP recommendations to minimize CDD/CDF formation and maximize their destruction. This paper summarizes recent and current activities associated with this study.

The utility of GCP for MWCs is in control of trace organic species; however, the impacts of GCP on other MWC emissions--e.g., NO_x, acid gases, and metals--have also been evaluated. GCP specifically addresses CDD/CDF since these compounds have relatively high toxicities when compared with other trace organics and have received a great deal of public attention in recent years. For many of the MWC regulatory alternatives under consideration, FGC equipment may be sufficient to ensure that CDD/CDF and other trace organics are removed from the stack gases and, therefore, will minimize health or environmental risk through air exposure. However, the September 1988 report of the Research Strategies Committee of the Science Advisory Board to the EPA Administrator identified as its number one recommendation that "EPA should shift the focus of its environmental protection strategy from end-of-pipe controls to preventing the generation of pollution" (7). This policy is also reflected in EPA's Pollution Prevention Policy Statement (8). GCP provides a mechanism whereby this recommendation can be implemented for trace organics control from MWCs. GCP will ensure that trace organics concentrations in captured fly ash are minimized, and will reduce air emissions during FGC system failure and/or bypass.

The basis for using GCP as a tool to control CDD/CDF emissions is the postulate that trace organic species are products of incomplete combustion. From fundamental considerations, all CDD/CDF and precursors can be destroyed if the necessary temperature and oxidizing environment are maintained for a sufficient period. The challenge of specifying GCP recommendations, then, is identifying design and operating conditions that will ensure uniform and stable combustion in MWCs.

BACKGROUND

ORIGINAL GCP RECOMMENDATIONS

GCP recommendations for control of trace organic emissions from mass burn waterwall, refuse-derived fuel (RDF) fired, and modular starved air MWCs were included in the combustion control volume of the Report to Congress (4). These recommendations, based on existing data and engineering judgment, specified quantitative or semi-quantitative design, operation/control, and verification limits for a set of MWC components deemed necessary to avoid excessive CDD/CDF emissions. The components addressed in the original recommendations included:

- temperature at fully mixed location
- air distribution/control
- auxiliary fuel
- turndown
- start-up procedures
- O_2 and CO in flue gas

Modifications and additions to the original GCP recommendations, being developed as part of the current study, arise from several considerations. First, the original recommendations addressed only three MWC technologies; the current analysis is directed toward eight MWC technologies:

- mass burn waterwall
- mass burn refractory
- mass burn rotary waterwall
- mass burn rotary kiln
- modular excess air
- modular starved air
- RDF spreader stoker
- fluidized bed combustor

Each of these MWCs incorporates distinct design and/or operating features that warrant technology-specific assessment. Secondly, a somewhat better understanding of the GCP components specified in the original recommendations and their effects on CDD/CDF emissions has been developed. This is attributable to an increase in the amount and quality of available data, as well as an expansion in analytical procedures allowing the full range of postulated CDD/CDF formation and destruction mechanisms to be considered. Finally, recently acquired field test data provide evidence supporting the need for two new GCP components--downstream temperature and particulate matter carryover.

CDD/CDF FORMATION

Numerous theories have been proposed to account for CDD/CDF emissions from MWCs, but no consensus has been reached regarding their validity in actual MWCs. The theories that have been proposed can be grouped into three general categories, as depicted in Figure 1.

The first explanation developed to account for MWC CDD/CDF emissions was that CDD/CDF present in the waste feed passed through the combustor undestroyed. Its original basis was the observation that some wastes contain CDD/CDF (9). However, the mechanisms in this category are no longer widely supported since many wastes have insufficient CDD/CDF contents to account for observed emissions, and CDD/CDF have been found to decompose rapidly at temperatures above 1110°F (600°C) and are, thus, unlikely to escape destruction in MWCs (10, 11).

Numerous in-furnace CDD/CDF formation mechanisms have been proposed to account for MWC emissions, including gas-phase and gas/solid reactions involving chlorinated hydrocarbons or unchlorinated hydrocarbons and chlorine donors (12-19). CDD/CDF synthesis is thought to occur in the burning refuse bed or in fuel-rich regions of the furnace. Specific compounds considered likely precursors include chlorobenzenes, chlorophenols, polychlorinated biphenyls, and polycyclic aromatic hydrocarbons. Primary decomposition of the miscellaneous materials present in municipal waste--paper, yard wastes, food wastes, plastics, wood, textiles, rubber, leather, and other organic materials--accounts for the presence of precursors in MWCs.

Bench-scale experiments by several investigators have provided convincing experimental evidence that CDD/CDF can be created in MWC systems downstream of the furnace (11, 13, 15, 20). Several field data sets further support the findings of these bench-scale studies. This third source of CDD/CDF is believed to result from catalytic reaction of appropriate precursor materials on the surface of fly ash. The reactions which create CDD/CDF under these conditions occur over a temperature range of approximately 480 to 660°F (250 to 350°C). Below 390°F (200°C), no reactions are observed to occur. Above 750°F (400°C), CDD/CDF and their organic precursors are subjected to thermal destruction. At temperatures from 480 to 660°F (250 to 350°C) and in the presence of concentrations of water vapor, oxygen, and hydrogen chloride, which are typical of MWC flue gas environments, carbonaceous material, most likely in the form of benzene ring structures, is converted to CDD/CDF. Experiments with MWC fly ash show that, in simulated MWC flue gas environments

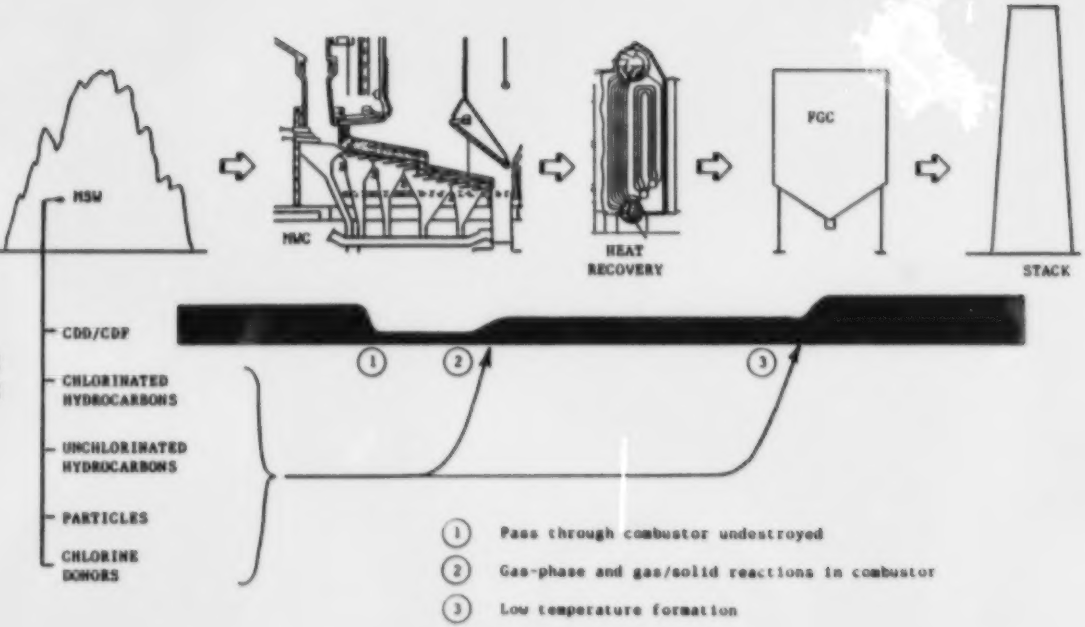
MSW
MWC
HEAT RECOVERY
FGC
STACK
CDD/CDF
CHLORINATED HYDROCARBONS
UNCHLORINATED HYDROCARBONS
PARTICLES
CHLORINE DONORS
①
②
③
① Pass through combustor undestroyed
② Gas-phase and gas/solid reactions in combustor
③ Low temperature formation

Figure 1. CDD/CDF Formation Theories.

at an optimum reaction temperature of 570°F (300°C), the amount of CDD/CDF on the surface of the fly ash can be increased seven- to ten-fold (15). The existence of this low temperature source of CDD/CDF emphasizes the importance of destroying all precursor materials (unchlorinated and chlorinated ring structures) along with any CDD/CDF before they leave the high temperature regions of the furnace.

GOOD COMBUSTION PRACTICE

The purpose of GCP is to identify design and operating conditions that ensure that trace organic emissions are minimized. Because of the various mechanisms by which CDD/CDF may be formed and the number of MWC design and operating parameters that influence formation, cause-and-effect supporting data for specific GCP components are quite limited. Rather, numerous bench-scale research and field test activities support the premise that low CDD/CDF emissions accompany good overall combustion. The current study has defined a set of design and operating components which, when properly specified and/or controlled, will result in good overall combustion.

Fundamental combustion theory dictates that, to effectively oxidize hydrocarbon species, a process must ensure that the fuel contacts the proper amount of air at a sufficiently high temperature and remains in contact long enough to allow completion of the oxidation reactions. An additional consideration for trace organic control is the low-temperature formation reaction discussed previously. The six major areas of MWC design and operation which impact these fundamental considerations are:

1. Temperature at fully mixed location
2. Amount and distribution of combustion air
3. Mixing
4. Particulate matter (PM) carryover
5. Downstream temperature
6. Waste feed

These six areas directly address the requirements of temperature, combustion air, and downstream formation. Reaction time is indirectly addressed by the temperature, combustion air, mixing, and PM carryover components. Gas-phase CDD/CDF and precursor destruction is essentially instantaneous at sufficiently high temperatures and local oxygen concentrations; i.e., the reaction rate approaches the kinetic limit. It is possible, however, that organic material could be in a solid phase—adsorbed onto an ash particle, as an unreacted piece of organic material swept through the flame, or as a condensed organic particle formed in fuel rich pockets in the lower furnace. In these situations the oxidation kinetics will not be "almost instantaneous" and, in fact, the process may be controlled by diffusion of oxygen into the reacting particle structure. This possibility is addressed by including PM carryover as a GCP component, with the objective of limiting the potential for organic and precursor material carryover with the particulate.

TEMPERATURE AT FULLY MIXED LOCATION

The importance of temperature in the formation and destruction of trace organic species is fairly obvious. To ensure that CDD/CDF are thermally destroyed in MWCs, it is necessary to maintain adequate temperatures in the presence of oxygen; i.e., waste thermal decomposition products must be mixed with sufficient combustion air. The minimum temperature at which thermal destruction of CDD/CDF occurs is determined by reaction kinetics. However, MWC flue gases also contain organic compounds that may be CDD/CDF precursors. Thus, the minimum temperature recommendation must be adequate to destroy both CDD/CDF and potential precursors to minimize post-flame formation. Two issues must be addressed in implementing the temperature recommendation: 1) the appropriate temperature to ensure destruction of CDD/CDF and precursor compounds, and 2) ensuring that all trace organic species are exposed to this minimum temperature.

Minimum Temperature Requirement

A ranking scheme developed by the University of Dayton Research Institute (UDRI) specifies thermal destruction temperatures for many of the RCRA Appendix VIII organic compounds (21). The 304 compounds in the ranking are grouped into nine classes based on the temperature at which 99 percent destruction of the compound is achieved in 2 seconds. The ranking is based on experimental data where available; otherwise, values were assigned based on similar compounds and/or reaction kinetic theory. The compound 2,3,7,8 TCDD has an assigned destruction temperature between 1580 and 1690°F (860 and 920°C). Several chlorobenzene species--which are suspected CDD/CDF precursors--have destruction temperatures between 1690 and 1830°F (920 and 1000°C). The UDRI data are based on non-flame, oxygen-starved conditions which are not representative of the conditions that would be experienced in an MWC flame; thus, these data are considered worst-case values for MWCs. However, in the absence of more directly applicable data, the UDRI data base provided the basis for the original temperature recommendation. Subsequent UDRI results indicate that destruction temperatures in the presence of oxygen may be 200°F (110°C) lower than those reported previously.

Parametric testing was conducted on the Pittsfield, Massachusetts, modular excess air MWC. At this facility, the secondary chamber outlets from three modular combustors (two operational and one standby) feed a single tertiary duct that supplies hot flue gas to two waste heat boilers (in parallel). Relatively low CDD/CDF levels were measured for numerous runs when tertiary duct temperatures were between 1450 and 1850°F (790 and 1010°C). Measured values increased substantially at temperatures below 1350°F (730°C). Based on the available measured data for this facility, it appears that low CDD/CDF emission levels can be achieved when operating the combustor at temperatures as low as 1450°F (790°C). This does not necessarily confirm that CDD/CDF is destroyed at this temperature, since there is no indication of the temperature profile or how the measured temperature relates to the minimum, average, and maximum gas temperatures in the duct.

Low CDD/CDF emissions have been observed at two fluidized bed combustor (FBC) MWCs. The Western Lake Superior Sanitary District (WLSSD) FBC co-fires RDF and sewage sludge, maintaining bed and freeboard temperatures of approximately 1475°F (800°C) and 1650°F (900°C), respectively. Northern States Power (NSP) co-fires RDF and wood waste in their FBC at LaCrosse, Wisconsin. Bed and freeboard temperatures in the NSP unit were maintained at about 1475°F (800°C) and 1500°F (815°C). FBCs have very uniform temperature and composition profiles; thus, the measured freeboard temperatures from these tests provide a relatively accurate indication of acceptable minimum temperature from a fundamental viewpoint or for systems without mixing limitations.

Temperature Variability

Gas temperatures at any point in the combustion system can vary considerably over time (temporal variation); in addition, measurements across a plane at a given location may also vary (spatial variation). Temporal variations coincide with short- and long-term changes in excess air levels and operating load. Spatial variations are largely the result of mixing profiles and heat transfer, where heat removal results in lower gas temperatures along the combustor walls than in the center of the plane of measurement. An example of this variation is illustrated in Figure 2, the temperature profile measured at the exit of the secondary combustion chamber at the starved air modular MWC at Charlottetown, Prince Edward Island (PEI). The profile indicates a gas temperature variation of about 235°F (130°C) between the wall and the center of the duct. The mass average temperature at this plane is approximately 1560 to 1650°F (850 to 900°C). The variation between temperatures at a similar plane in a waterwall facility may be even greater due to higher heat removal along the waterwalls.

COMBUSTION AIR

Three aspects of combustion air control are addressed by GCP: excess air levels, primary air control, and secondary air capacity and control. Excess air level is included as a GCP component since it addresses the overall air requirement of the system. The primary and secondary air components address zonal air requirements to meet local oxygen demands needed to avoid formation of difficult-to-destroy soot particles.

Excess Air

The amount of excess air introduced to the combustor must be sufficiently high to minimize the existence of fuel-rich pockets and sufficiently low to avoid quenching the combustion reactions. The range of excess air levels that will satisfy these objectives varies for each combustor technology. The upper end of the range is limited by the minimum furnace temperature that can be maintained at the fully mixed location. This value can be approximated for refractory wall MWCs using the relationship between excess air and adiabatic flame temperature, shown in Figure 3. The adiabatic flame temperature is less representative of the actual temperature in systems with a significant level of heat extraction prior to the fully mixed location (e.g. waterwall MWCs), but the figure provides an upper bound for estimating purposes. The lower end

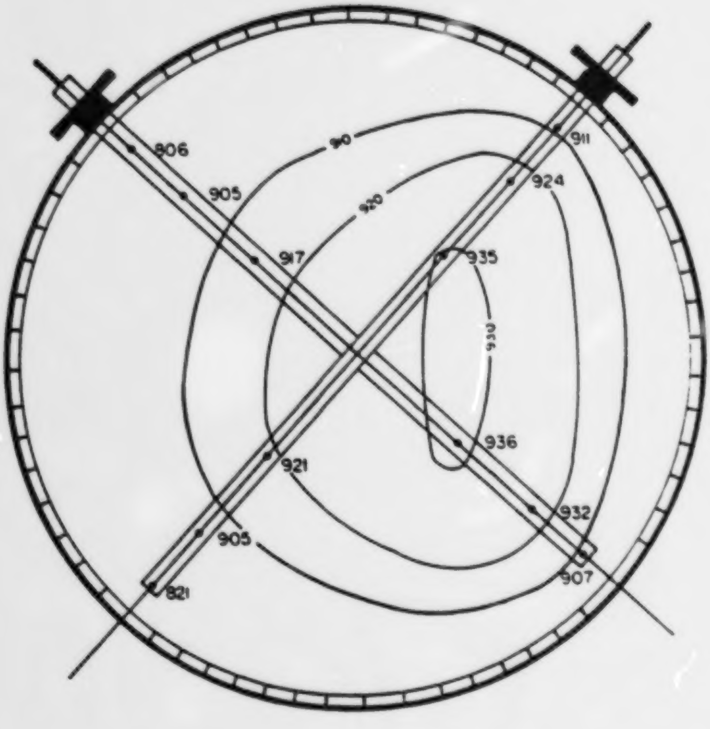
806
905
917
910
920
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Figure 2. Temperature distribution (°C) in Secondary Chamber Exit of Charlottetown, Prince Edward Island MWC.

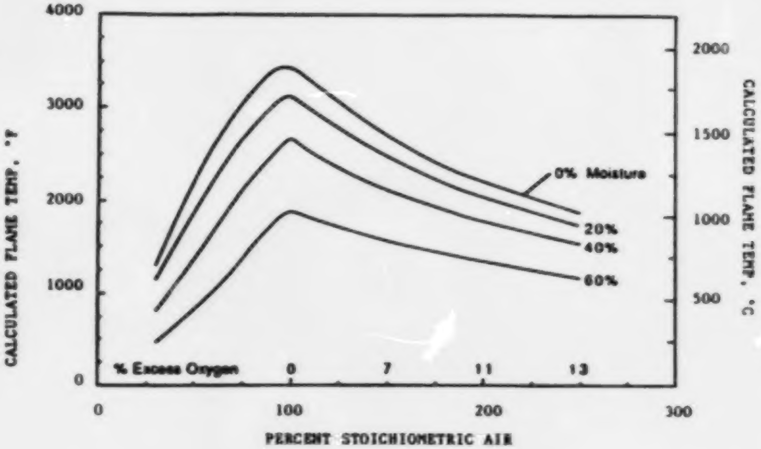

Figure 3. Relationship Between Adiabatic Flame Temperature and Percent Stoichiometric Air for MSW Fuel.

of the excess air range is defined by technology-specific mixing characteristics. Lower minimum excess air levels may be recommended for systems with inherently good mixing, such as FBCs. Other designs (e.g. mass burn waterwall) require greater quantities of excess air to ensure that all fuel-rich zones are adequately mixed with oxygen.

Three data sets which support the establishment of excess air limits for various MWC technologies are the result of parametric tests carried out at Pittsfield, Massachusetts (modular excess air), Quebec City (mass burn waterwall), and PEI (modular starved air). Results from these three programs are illustrated in Figure 4 and summarized below.

- Pittsfield - Figure 4 illustrates the effect of oxygen concentrations at the tertiary duct sampling location on CDD/CDF emissions. CDD/CDF emissions increase at both ends of the U-shaped curve, presumably due to insufficient mixing at the low end and quenching and/or higher PM carryover at the high end. A key objective of GCP is to specify excess air operating conditions that fall in the trough of this U-shaped curve.

- Quebec City - CDD/CDF emissions were measured over a wide range of operating conditions during the Quebec City parametric testing. Statistical analysis of the data was performed to evaluate the effects of operating parameters on CDD/CDF emissions. A fairly strong correlation was shown to exist between high excess air levels and CDD/CDF; however, this effect may also be attributable to PM carryover.

- Prince Edward Island - The relationship between excess oxygen and CDD/CDF emission levels at the stack of the PEI MWC is similar to the Quebec City correlation. Although the data indicate that lower emissions of CDD/CDF can be achieved through careful control of excess air levels, the base of the PEI curve corresponds not only to a lower excess air level, but also to a higher operating temperature. Therefore, the lower emissions may be the end result of several parameters acting in unison to provide the conditions specified by GCP.

These data sets do not provide the basis to establish quantitative excess air limits associated with GCP. Rather, meaningful limits can only be established on a case-by-case basis since so many other factors may influence the relationship between excess air levels and the limits of good combustion performance.

Primary Air Control

The proper amount and distribution of primary (underfire) air is essential in maintaining bed burning stoichiometry and good waste burnout. The GCP recommendations for primary air control must be applied in a technology-specific manner according to the design objectives of each combustor type. For example, in conventional mass burn waterwall MWCs the objective is met by

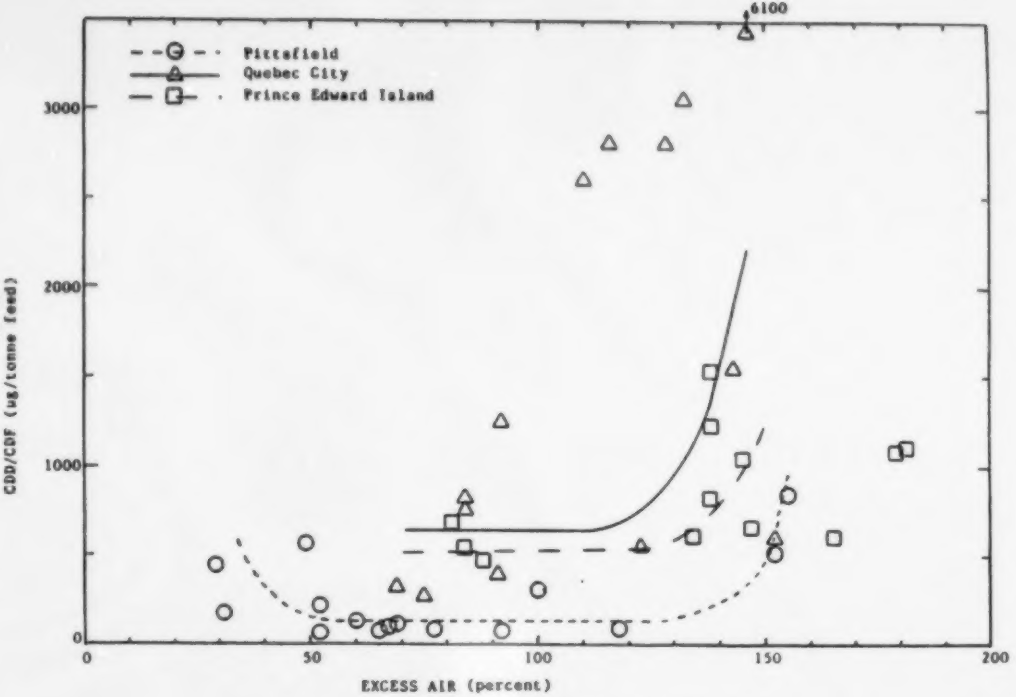

Figure 4. Influence of Excess Air on CDD/CDF Emissions.

the use of multiple underfire air plenums, each with separate controls. The distribution of underfire air to each grate section is varied to accomplish drying, burning, and burnout of waste. In a starved air MWC, the requirement is met by controlling the amount of air supplied in the primary combustion chamber so that substoichiometric conditions are maintained at operating temperatures low enough to prevent clinker formation. RDF spreader stoker boilers can be designed to meet GCP goals either with multiple underfire air plenums similar to mass burn waterwall MWCs, or with metered feeding systems that provide a uniform distribution of waste feed on the traveling grate, thereby maintaining desired air-to-fuel ratios and minimizing channeling of underfire air through less dense regions of the fuel bed.

The best full-scale emissions data set that provides support for selection of primary air distribution as a component of GCP comes from the Quebec City parametric test. The existing underfire air arrangement (two plenums) was modified to include a 3x2 plenum arrangement beneath the burning grate. Prior to the retrofit, very large swings in air flow (20-60 percent normally, and as high as 0-100 percent) were required to maintain the desired steaming rate, resulting in poor distribution, channeling, and poor control. The modification corrected these problems and contributed to the reduced air emissions reported previously for this unit. The lowest stack emissions were measured when the primary air system supplied 60 to 65 percent of total air.

Secondary Air Capacity/Control

The most common method used to achieve good mixing in MWCs is properly designed secondary (overfire) air systems. The design of the secondary air system to provide mixing through penetration and coverage of the furnace cross section must take into account nozzle sizes, velocities, and arrangement as well as furnace geometry and thermodynamics.

Very few emission testing programs have evaluated the effects of overfire air design and operation on emissions of CDD/CDF. The Quebec City MWC retrofit included a redesign of the overfire air system, and the result of the program was a reduction in CDD/CDF of nearly two orders of magnitude. Similar results were achieved in a retrofit at the Hampton, Virginia, MWC following a system retrofit that included reoriented overfire air nozzle rows.

In-furnace temperature, CO, and velocity profiles were measured in a European mass burn waterwall MWC while operating with variable combustion air distributions. Flow modeling studies were also used as a diagnostic evaluation. The test results showed that mixing is enhanced by the proper distribution of secondary air, and that furnace velocities and residence times are also improved. No CDD/CDF measurements were made as part of this evaluation.

The first phase of a gas reburning demonstration conducted at Riley Takuma's Olmstead County MWC also addressed the use of overfire air on pollutant emissions. Operation of the overfire air system had the most significant effect of any parameter evaluated on the control of NO_x, CO, and unburned hydrocarbons. A major conclusion of the program was that an inverse relationship exists between CO and NO_x as a result of the excess air operating rate, which is controlled by adjusting overfire air flow rates. The results conclusively showed that hydrocarbons vary inversely with overfire air flow rate.

MIXING

Intimate mixing of fuel and air is widely recognized as a requirement for complete combustion. Poor mixing can shift the rate-limiting step for CDD/CDF and precursor destruction reactions from a kinetic limit to a mixing limit. Good mixing depends on introducing the proper amounts of air at the proper locations and ensuring that the air thoroughly contacts the solids and/or gases present at and downstream of the point of introduction. Three design and operating parameters that impact the ability to achieve good mixing are combustion air control, load (turndown), and combustor configuration. Combustion air control was discussed above; the effects of operating load and combustor configuration are detailed in the following paragraphs.

Operating Load (Heat Release Rate)

All MWCs have a rated capacity that is generally established based on a volumetric heat release calculation and expressed in tons per day of feed and/or pounds of steam per hour at specific pressure and temperature. The furnace temperature profiles and mixing patterns are affected to a large extent by the operating load. As the waste feed rate is reduced, the total air flow required to complete combustion is likewise reduced. As stated previously, failure to maintain a specified excess air level may result in quenching of the combustion reactions. However, the importance of both primary and secondary air flows has also been mentioned. A reduction in excess air at a given primary/secondary air ratio will require that both primary and secondary air flows be reduced. There is a point at which the reduced secondary air flow rate is insufficient to maintain penetration and coverage, limiting the level of mixing that is achievable.

Increasing firing rates above maximum rated capacities can also affect temperatures and mixing patterns. As heat input rates increase, total air flows and vertical velocities will also increase, giving rise to quenching of reactions, disruption of mixing patterns, and increased PM carryover. Operational problems related to flame impingement, erosion, and fireside corrosion also are limiting factors associated with excessively high loads.

The effect of operating load on CDD/CDF emissions can be seen in the results of parametric tests at the Quebec City mass burn waterwall MWC. The data, shown in Figure 5, include results from 10 test runs performed at "good operating conditions" which are defined based on air distributions and operating temperatures. Low load (70 percent) and high load (115 percent) emission rates are both higher than those measured during operation at 100 percent load.

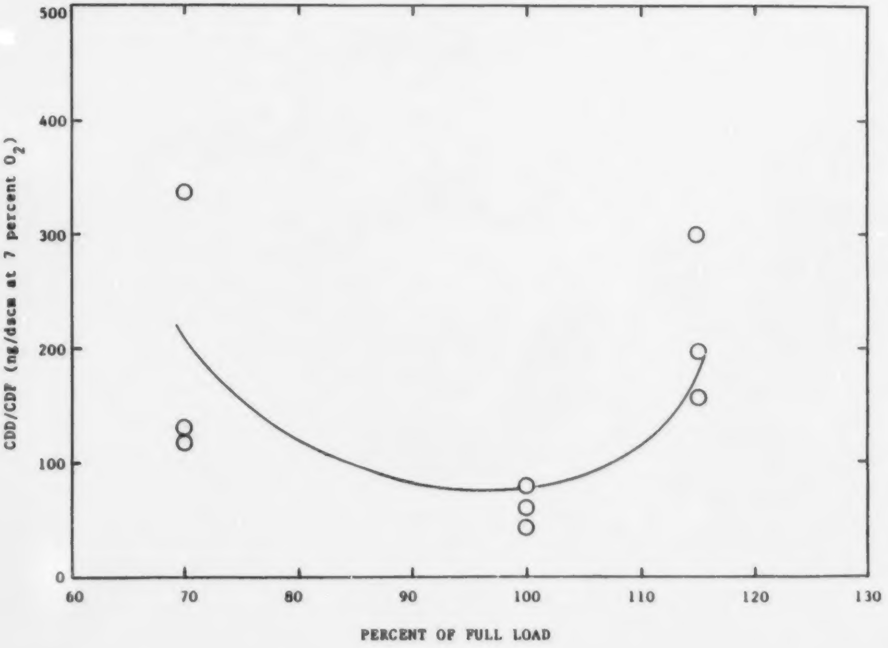

Figure 5. Effect of Load Variation on CDD/CDF Emissions.

Combustor Design

Mixing can also be effected through combustor design measures; mixing is effectively correlated with turbulence of the combustion product flow field. Changing the flow direction of either part or all of the flue gas stream will promote turbulence. Conversely, straight, smooth flow paths with constant cross-sectional area tend to have more laminar flow characteristics. Combustor design measures that enhance mixing include bull noses, baffles, turns, and changes in duct shape or cross-sectional area. Assessment of this GCP component has been primarily qualitative as there are very limited data which allow direct evaluation of the impacts of design modifications intended to promote mixing.

PARTICULATE MATTER CARRYOVER

The amount of particulate carryover from an MWC can affect emissions of metals and organic pollutants. Many potentially toxic metals are present in municipal solid waste and may be emitted as a fraction of total particulate matter. In addition, correlations have been established between the level of PM emitted from the boiler of a mass burn waterwall MWC and the CDD/CDF emission levels in the stack. The current thinking regarding this relationship is that the particulate matter provides a surface area where catalytic reactions occur which result in formation of CDD/CDF. Alternatively, particulate matter that is swept out of the high temperature combustion zone before organic matter is thermally destroyed will provide available organics for low temperature formation of CDD/CDF.

PM carryover is coupled with the combustion air components (excess air, primary air, and secondary air). High excess air levels will normally increase PM carryover, as will a high primary-to-secondary air ratio. The unabated PM emission level from an MWC also varies according to combustor design and operation, and in some cases, waste physical properties. For example, RDF spreader stokers exhibit higher PM carryover than many other designs due to the use of a semi-suspension firing mode. Conversely, starved air modular MWCs are designed with low air flows and velocities in the primary combustion chamber, and typically exhibit relatively low PM carryover.

As stated previously, the Quebec City parametric test included emission testing at multiple combustor operating conditions. Single parameter correlations were evaluated for a range of variables. The best single parameter correlation ($r^2 = 0.876$, critical $r^2 = 0.306$) was a comparison of uncontrolled ash (PM) entering the ESP versus CDD/CDF in the stack. Two other variables with extremely good single parameter fits were flue gas flow rate ($r^2 = 0.771$) and primary air flow rate ($r^2 = 0.723$).

RDF spreader stokers as a class have typically exhibited higher levels of CDD/CDF emissions than many other MWC technologies. It is suspected that PM carryover may be a major contributor to this observed trend. One recent set of emissions data obtained during field tests conducted by the Gas Research Institute confirms the results obtained at Quebec. Correlation of PM versus stack CDD/CDF for both data sets results in a single parameter fit of $r^2 = 0.899$.

DOWNSTREAM TEMPERATURE

Several full scale emission testing programs have produced results indicating that, under certain conditions, CDD/CDF concentrations in MWC flue gases increase between the combustor exit and the stack. A number of bench scale experiments carried out with MWC fly ash samples indicated that CDD/CDF concentrations increased dramatically when the fly ash sample was heated in the presence of an oxygen-rich gas stream (12, 16). The amount of CDD/CDF formation varied based on retention time and temperature, with the greatest formation occurring at 570°F (300°C). At temperatures below 480°F (250°C) and above 660°F (350°C) the rate of the formation reactions was considerably diminished. Although complete elimination of the formation reactions may not be possible because the flue gases must pass through this temperature window, the conditions thought to result in the highest rate of CDD/CDF formation (i.e., extended solid and gas residence times in the temperature window) can be minimized.

At least two facilities (Lawrence, Massachusetts, and Hampton, Virginia) have test data comparing CDD/CDF stack emissions prior to a retrofit, when ESP operating temperatures were in the 500 to 600°F (260 to 315°C) temperature range, to data measured following the combustion retrofit, when ESP temperatures have been reduced to less than 450°F (230°C). For both, the stack emissions were reduced by more than an order of magnitude from the pre-retrofit tests, although other aspects of the retrofits may have also contributed to the lower emissions.

Three similar mass burn waterwall combustors (Westchester, New York, North Andover, Massachusetts, and Pinellas County, Florida) have conducted simultaneous ESP inlet and outlet sampling for CDD/CDF. The North Andover and Pinellas ESPs were operated at temperatures between 500 and 600°F (260 and 315°C). The Westchester ESP operated at temperatures between 430 and 470°F (220 and 245°C). Increased concentrations of CDD/CDF as high as 100 percent above ESP inlet levels were measured at North Andover and Pinellas. Stack concentrations at Westchester were lower than ESP inlet concentrations for every test run.

A qualitative evaluation of all existing CDD/CDF emissions data provides additional evidence supporting the downstream formation mechanism. Nearly every MWC in the emissions data base with CDD/CDF stack emissions greater than 1500 ng/dscm has a flue gas cleaning device that operates in the temperature range where CDD/CDF formation has been observed.

WASTE FEED

Combustion stability can be greatly affected by changes in waste feed density, feed rate, and heating value. The uniformity of waste feed is generally more important in FBCs and RDF spreader stokers. This is well demonstrated by two data sets generated at a circulating fluidized bed RDF facility in Sweden, shown in Figure 6. The top data set indicates continuous CO emissions measured in the stack while firing an RDF that was processed with a single-stage shredder. There are frequent CO spikes in excess of 2000 ppmv during a period of just over 1 hour. The second test was conducted while firing RDF which had been processed with two stages of shredding, resulting in a more uniform waste feed with smaller nominal particle size. The CO histogram shows that the majority of spikes are eliminated and combustion conditions are far more stable, resulting in a far lower potential for trace organic emissions.

Additional supporting data are available from an RDF spreader stoker tested by the Gas Research Institute. Spatial variations in temperature were observed at various sampling locations in the furnace. However, temporal variations dominated spatial variations; temperature fluctuations of more than 300°F (165°C) were observed over a 10-minute period. The temporal variations correlated with RDF feed uniformity. Periodic slugs of RDF feed resulted in significant changes in temperature at a single point, and these instabilities could be one major cause of higher CDD/CDF emissions.

CONCLUSIONS

The data examined during this study support the premise that trace organic emissions can be controlled using design and operating measures specifically designed to ensure good overall combustion. Quantitative and semi-quantitative recommendations for each of the GCP components are being developed by ORD. If GCP components are included in the MWC standards and guidelines scheduled for proposal in November 1989, they will be based on the considerations discussed in this paper.

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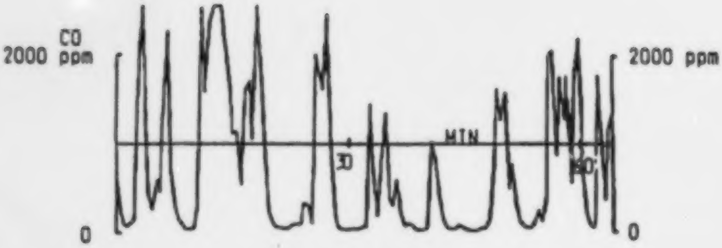
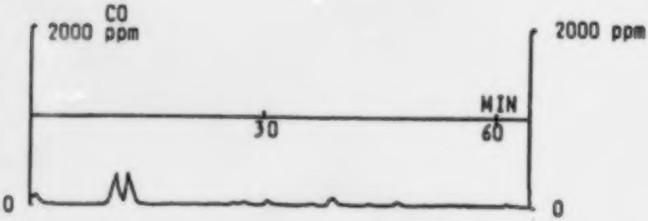


A. Single-stage shredding.



B. With secondary shredding.

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THE NIMBY SYNDROME IN MUNICIPAL WASTE MANAGEMENT

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ABSTRACT

The NIMBY Syndrome is a convenient generic label for a complex array of social forces and community concerns. One distinctive feature of NIMBY, is its ability to bring project siting to an abrupt halt.

With waste management needs approaching crisis deadlines and facility siting becoming increasingly difficult, municipalities must anticipate the inevitability of a NIMBY response. To develop the tools to alleviate NIMBY reactions, it is time to carefully examine some of the forces fueling the public's concerns. With a more sophisticated understanding of NIMBY, municipalities can then begin to initiate policies that will address the community's concerns.

What do prisons, incinerators, group homes, new highways and sewage treatment plants have in common? - United and too often successful public opposition to their siting. Where is the source of the problem in what has essentially become a serious stalemate between essential services and community hostility? Can you blame poor planning, insufficient institutional requirements or irrational citizens? No - I don't believe you can say that the planners aren't doing their job or the regulatory framework is too relaxed or the citizens are without some substantive concerns for the proposed project.

I think we are misdirected if we single out the "NIMBY" Syndrome as the cause of abandoned project sitings. The problems to siting proposed projects is far more complex than the organized public opposition now commonly labelled the "NIMBY SYNDROME". Public opposition has in many cases become an insurmountable force in waste management planning. Effective NIMBY activities have emerged from a complex series of problems with and reaction to current siting practices. Unless we find some way to counteract the effects of successful site opposition, then we will continue to have to rely on outdated, overworked technologies and waste facilities that reflect the technological expertise we relied upon twenty and thirty years ago.

Municipal garbage, be it residential or commercial, is not going away. The need to push waste management plans through planning, approval and construction is now often responding to crisis needs. Many municipalities do not have the luxury of years to replace their current waste management infrastructure. Working against these impossible deadlines is the growing public consciousness and priority on the "environment".

How then do we approach the current state of the NIMBY syndrome? I ask that you start with certain basic beliefs. First, you must take the public seriously. I believe that most people in opposition have a sincere, substantive basis for voicing their objections. To treat them as irrational or poorly informed is to do the 1980's public a great injustice.

Second, in the planning of any waste management facility, take as a given that it will generate a significant NIMBY response. Don't focus your energies on whether or not the criticism is valid. Redirect yourself to finding the root cause of the criticism and what the proponent municipality can do to address it.

Finally, don't ever initiate a siting exercise with the naive hope that any problems you encounter are nothing more than disagreements that can be resolved through dialogue. As I will discuss below, NIMBY is a convenient title for a complex series of issues, fears and forces at play in today's communities. If you understand the components then you set the framework for changes designed to alleviate the current stalemate.

SOURCES OF NIMBY

(1) CREDIBILITY AND TRUST

Several authors have identified the public's lack of trust in both governments and experts as a source of NIMBY[1]. Related closely to that lack of trust is the credibility of those who speak out in favour of a particular waste management project. How the information is received by the public will relate closely to the credibility and trustworthiness of the source. I don't believe you can underestimate the importance of credibility and trust when discussing NIMBY. Unfortunately, trust is more easily lost than won.

Why then does it appear that government has lost the people's trust? I believe the public holds government responsible for irresponsible environmental management practices. Communities living with problem waste management facilities see their government representatives as having failed to provide the necessary regulatory safeguards.

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[1] Michael E. Kraft and Bruce B. Clary, 'Assessing Citizen Participation in Environmental Policy: The NIMBY Syndrome and Radio-active Waste Disposal", (delivered at the 1988 Annual Meeting of the American Political Science Association; C.P. Wolfe, 'The NIMBY Sydrome: Its Cause and Cure'. New York: Social Impact Assessment Centre; Audrey Amour, 'Social Dimensions of Facility Siting' from Hazardous and Toxic Materials: Safe Handling and Disposal, 2nd ed. (Howard Fawcett, editor); Audrey Armour, 'Is Faciltiy Siting a No-Win Situation?' Presentation to 'Environmental Conflict Resolution', The Banff Centre School of Management; E. Peelle and R. Ellis, 'Hazardous Waste Management Outlook: Are There Ways Out of The 'Not-In-My-Backyard' Impasse?' Forum for Applied Research and Public Policy, Sept. 1987.

Government must now rebuild that trust in part through the development of greater performance requirements. The Canadian Federal Government has identified priority pollutants for immediate action. The Ontario Ministry of the Environment is imposing increasingly stringent discharge guidelines on everything from sewage waste to smoke stack emissions. The community must believe that their government is imposing real, effective environmental safeguards if we expect to win back their trust.

If the public doesn't believe in the standards being set by government, then the 'technical experts' gain no credibility by meeting and exceeding those standards. I am all too familiar with the public's dissatisfied reaction when they are given guarantees by the technical consultants that a proposed municipal incineration facility not only meets but exceeds all government guidelines regarding emissions. Regrettably, state-of-the-art technology does not have a history in use to generate its own credibility.

Even if current regulatory standards represent defensible standards for public health and safety, many still criticize current enforcement strategies. If you can negate the fear that enforcement is weak and ineffective then you can reduce the public's belief that it is up to them to protect themselves. NIMBY is both a proactive mechanism to offset the belief that current government programs do not provide the necessary protections for people and the environment and a reactive emotion stemming from past practices.

(2) EQUITY

No lawyer's presentation would be complete without reference to the principle of equity. The idea of equity is not new to the discussion of NIMBY[1]. Equity takes into account the distribution and impact of risks (to be discussed later) as well as the perspective of host communities on the siting of municipal waste facilities.

The reality of current siting strategies is that "preferred sites" will almost inevitably be located in a largely rural setting. NIMBY reactions are thereby fueled by the host community's perception that an urban problem is being imposed on its rural 'neighbour'. Take for instance, the proposal presented to Metropolitan Toronto's politicians in September of 1988, to haul their garbage by train to remote parts of Northern Ontario.

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[1] Op. Cit. Wolfe, C.P.

The perception you create is an inequitable shift in responsibility away from the group that created the bulk of the problem. In terms of a basic cost/benefit analysis, the host community is left with the perception of intangible benefits and high disadvantages. They live with the risk (be it real or perceived) of the facility, while the generators experience no direct social impact from their waste practices.

To offset these concerns, municipalities must search for a fairer apportionment of benefits and risks. Negotiated strategies and proposals perceived by the host community as 'benefits' may shift the balance of equities and avoid new site selections.

(3) LIFESTYLE AND HEALTH CONCERNS

Lifestyle factors influencing the NIMBY response include perceived impact on the routines of the host community, effects on property values and the emotional and physical stresses related to living near a waste facility. This perceived impact on health and lifestyle is perhaps the most emotional component of NIMBY. As such, it is also the most difficult to address in the form of a solution.

Added to the issue of lifestyle is the public's heightened health consciousness[1]. We have noted a dramatic increase in concern over avoidable health risks as divergent as overexposure to the sun, steroids in beef cattle and spraying of fruits and vegetables. This general trend also impacts on the public's willingness to accept a "voluntary risk" - namely a waste management facility.

Therefore, in dealing with NIMBY you must acknowledge both the quantifiable concerns for property values and economic loss and the intangible sanctity of community cohesion, health impacts and the social fabric of the host neighbourhood.[2]

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[1] Op. Cit., Audrey Armour 'Is Facility Siting a No-Win Situation?'

[2] Op. Cit., Wolfe, C.P.

(4) FEAR OF RISKS

At the root of perceived risk is the insurmountable fact that analysis of state-of-the-art facilities will be limited so long as they have limited practical application. People want to see operating data not laboratory statistics. "Models" or small scale "pilot projects" do not provide the necessary reassurances of safety.

Risk analysis, from the community's perspective is motivated by four key factors:

(1) fear of accidents, malfunctions and exposure to toxins;

(2) distrust of the industry and regulators responsible for monitoring compliance;

(3) negative impacts on property values, quality of life and the aesthetics of the community; and

(4) belief that experiences at older, existing facilities will be translated to the proposed facility.[1]

In addition, the community's perception of risks associated with a facility relate to certain risk management practices, and whether or not they believe them to be effective. Michael Elliott Poirier in his paper[2], emphasizes prediction, prevention, detection and mitigation as forces in the issue of risk and risk management.

Prediction is the ability and skill to forecast system problems. Prevention includes the technical expertise to design, build and manage a reliable system. Detection addresses the concern for safeguards built into the facility to quickly detect hazardous conditions. Mitigation reviews the difficult issue of reversing the damage once it is done.[3]

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[1] Michael Poirier Elliott, "Improving Community Acceptance of Hazardous Waste Facilities through Alternative Systems of Mitigating and Managing Risk" in Hazardous Waste Volume 1, Number 3, 1984.

[2] Ibid, Elliott, Michael Poirier.

[3] Ibid, Elliott, Michael Poirier.

Intertwined with the components of perceived risk are the distinct perspectives with which risk is assessed by the public and the technical consultant. The community will base their risk perception on what Mr. Poirier refers to as 'impressionistic reasoning' while the technical experts will base their perceptions on 'scientific reasoning'[1].

I believe it is essential to understand the different approaches each party takes in assessing risk if you are going to develop appropriate mitigating measures. The host community may respond to social control measures such as monitoring programs, contingency plans, citizen advisory boards, financial securities and information exchange networks. It is a mistake to focus exclusively on engineering protections and 'back-up' systems when attempting to reduce the impact of risk perceptions. Scientific measures are only effective if the perception of risk is based on scientific reasoning.

(5) TECHNICAL RATIONALITY vs. SOCIAL RESPONSIVENESS

The Ontario waste management planning process (like most of its counterparts) begins with the presumption that scientific, technical criteria is the best, most defensible basis for site and technology selection. This foundation of technical rationality is the cornerstone of our municipal waste management planning process and our environmental assessment responsibilities. Our technical consultants (and obviously our lawmakers) believe that technical criteria and decisions result in objectively sound choices.

What many community organizations have discovered is that technical decisions are often the product of the 'experts' subjective judgments. Analysis of sites is often limited to what can be quantified, leaving values, insights and public attitudes at a disadvantage. Technical rationality constrains social responsiveness if it is held out as the only measure of a good site.

Starting the planning process with the presumption that technical, quantitative criteria are objective and sound is more likely to lead to recommendations that are held suspect by the impacted community. A 'technically sound' choice will be of little comfort to those people who believe their concerns are secondary. Strike a compromise somewhere between allowing NIMBY to drive your project and treating the public's role as minor in comparison with your "experts".

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[1] Op. Cit. Elliott, Michael Poirier.

(6) PUBLIC PARTICIPATION AND PUBLIC RELATIONS

When discussing public participation programs and public relations techniques one basic problem surfaces repeatedly - they start too late in the planning process. If the public believes that decisions have already been made then you, as a promoter of the project, are already at a significant disadvantage. If you then proceed to 'hard sell' those decisions or guarantee the public they still have an opportunity for meaningful input, I believe you are only stoking the fires of discontent.

A NIMBY reaction will be encouraged if the public perceives that they have lost an opportunity to effect change. When the community feels disempowered and confronted by the promoter's choices then the instinctual reaction will be to dig their heels in and fight.

In addition to the impact of delayed public participation there are certain public relations strategies that influence NIMBY reactions. There is a tendency on the part of the facility promoter to over-emphasize their own point of view. As part of the "sales pitch", the alternatives to the undertaking tend to be underplayed.[1]

Today's sophisticated public feels the impact of a narrowly focused approach to selling the proposed facility. If the public believes that the planning has narrowed too quickly to a particular end, then this strategy coupled with delayed input promotes a NIMBY response.

A second public relations strategy that causes problems is the tendency to justify decisions and recommendations in terms of their larger public policy and societal impacts.[2] This strategy emphasizes the good to the many and underrates the perspective of local citizens. Any public relations strategy designed to push the greater good to society as a whole is insulting to the host community and counter-productive to reducing NIMBY.

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[1] Audrey Armour, "Resolving Facility Siting Conflicts", Faculty of Environmental Studies, York University.

[2] Op. Cit Armour, Audrey, "Resolving Facility Siting Conflicts".

(7) PUBLIC'S ABILITY TO COMPREHEND

Site proponents may too often underestimate the public's ability to comprehend technical information. To make the mistake of presuming scientific ineptitude will encourage a strong reaction from the public you have insulted by that presumption.

The general growth in environmental awareness has had a direct impact on the public's willingness to comprehend and their patience in the face of detailed information. Most communities are no longer satisfied with the advice of 'experts' and expect to be given the opportunity to understand the information first-hand. To deny the public that opportunity heightens the NIMBY response.

In my experience with the public, their is a fine line between information that is too sophisticated in its language and reports that are so simplified as to insult the 'average' reader for whom they were written. The answer? Presume your public has the capacity and/or willingness to absorb every level of information at your disposal. Don't start with the expectation that your experts will have to write 'down' to their audience. Let the information provided and any explanations that are given be dictated by the requests you receive. The public is more than willing to set the standard of the information exchange.

(8) ADMINISTRATIVE HEARING AS DISPUTE SETTLEMENT TOOL

Both the legalistic mind-set and regulatory framework in Ontario (as elsewhere) place too much reliance on the administrative hearing as an arbiter of unsettled disputes. The fact that these hearings have become quasi-judicial, governed by formal rules of practice and couched in legal language is antithetical to 'co-operative, consensus-building exchanges.'[1] Parties find themselves trying to communicate effectively and resolve concerns in a 'we-they' forum.

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[1] Op. Cit. Peelle, E. and Ellis, R.

A proponent should not take the position that they can abandon their mediation efforts in the early planning process and leave the difficult decisions to an independent tribunal. In an ideal world, the planning and development stages will be synonymous with consensus-building, mediation and dispute-resolution. The administrative hearing would then be left to resolve a narrow set of outstanding issues and affirm the decisions reached between the parties. NIMBY activities will be positively impacted by a shift in emphasis toward self-directed settlement strategies and away from reliance on administrative boards.

REMEDIAL MEASURES

I can't possibly claim to have reviewed all of the component sources of NIMBY. I do believe, however, that the reader has a sufficient overview to pass judgment on some of the remedial measures set out below.

(1) PUBLIC EDUCATION PROGRAM

To my knowledge, no one has mounted an extensive public education program as a first step to site selection. While many would criticize the time and money expended to bring citizens to a certain level of technical expertise, you can also argue that the public has a right to be provided with the necessary tools to contribute to the decision making process.

If we believe that the incinerator technologies being promoted will survive the intensive review of an administrative tribunal, then surely it can withstand the scrutiny of an educated public. It might even be argued that an educated public might be more inclined to reach a decision that is less emotional in nature and more detailed in its technical commentary.

The form of public education is quite flexible. It may be dictated by the size of the audience it intends to reach. There is no reason in my mind why the public education process can't start as soon as the municipality has identified a need. The review of that need and the alternatives available to meet the municipality's particular waste quantities may be presented in something as simple as a mass circulated pamphlet.

I can't stress enough how strongly I believe that early education and wide dissemination of information is a crucial starting point. Widespread public education leads the citizens to believe they are being kept well informed. An exhaustive supply of information ranging widely in its level of detail and sophistication allows the public to access the resources they need to refine their own levels of knowledge. The municipal Councils making waste management decisions are generally no better equipped to make those decisions than any other member of the public and all are equally entitled to the resources to refine their knowledge.

(2) IMPACT MANAGEMENT OPTIONS

Development of generic impact management options is another pre-siting strategy to reduce NIMBY. You gain credibility by developing the impact management package before you face an identified hostile community. The options should be a series of safeguards and mitigation measures that can be negotiated with particular host communities.

There are a number of divergent offerings that can be included in your impact management package. I don't want to leave the impression that they are all necessary to reduce NIMBY reactions. Which options are actually chosen will probably reflect the special characteristics of the particular host community.

The following are a list of suggested impact management options.

a) Establish contingency accounts with designated funds to cover different aspects of the facility. One account may exist for problems arising during construction while a separate account relates to occurrences during management of the facility.

b) Payments may be made to the municipality in the form of a royalty or tipping fee per tonne of garbage disposed of at the site.

c) Whether the facility is run privately or publicly, there may be an agreement to direct a portion of the tipping fee to redevelopment projects within the host community.

d) There may be agreement to employ a set number of local residents at the facility. At what level and in what capacity may be the subject of specific negotiations.

e) Depending on certain factors (such as the proximity to residential housing or the size of the facility) agreement may be reached to fund the relocation of certain residents who meet a particular impact formula.

f) Agreement may be reached whereby the facility operator will fund a series of air, water, soil and/or health monitoring programs.

The options in a generic impact management package are only limited by your imagination. Kent E. Portney in his 1984 study[1] made offers which included paying all property taxes for 10 years, repaving the host community's streets, awarding scholarships to local high school seniors and buying each family an insurance policy against the financial losses or health risks resulting from an accident.

To put the package on the table before siting has been carried out avoids any impression of a particular bias against the host community. It is valuable for the community identified for a new waste management facility to understand beforehand what range of impact management options are available for negotiation. This avoids a two-tiered battle with the project proponent. You eliminate the battle over what options may be negotiated and concentrate on the finer points of the generic options that have already been identified. This has the distinct possibility of reducing points of conflict for the parties.

(3) PUBLIC PARTICIPATION

I have already alluded to the frustration people feel in being included in the public participation process only after decisions have been made. While most of those decisions are not cast in stone, the process, having been carried out without public dialogue, limits the credibility of any participation programs, undermines trust and serves as a catalyst for NIMBY.

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[1] Kent E. Portney, 'Allaying the NIMBY Syndrome: The Potential for Compensation in Hazardous Waste Treatment Facility Siting' in Hazardous Waste, Volume 1, November 3, 1984.

Just as I believe that public education should be an early priority, so too I am convinced that it is never too early to establish meaningful public input programs. However, there is also more to public participation than when you initiate the program.

Part of the emotions behind NIMBY are feelings of powerlessness and insignificance. You counteract those feelings by giving the public input into the development of the planning structure, collection of data, monitoring of the consultants and evaluation of recommendations. The municipal proponent must be seen to have created extensive interactive communication channels. Your planning process must be such that the public sees that they have an opportunity to affect that process and change the course of study if valid information comes to light.

In addition to formulating detailed participation programs, I believe that the legislators have an obligation to formalize the public's right to participate. Currently, under Ontario Environmental legislation, public participation and information access is not formalized in the regulatory framework. Yet, every Environmental Assessment Board decision devotes considerable attention to the quality and quantity of public input into the proponents project.

Without some basic legislative standards for public participation, the public feels they have no guarantees of when or even if they will be heard. Granted, for a municipal proponent it would be political suicide to undervalue the participation component of your facility siting program. However, there are few legislated or practical standards against which you can evaluate your efforts and more importantly, against which the public can measure your performance.

(4) PUBLIC PARTICIPATION FORMATS

The classic mistake that is made by municipalities is the belief that "town-hall" type, large scale meetings are an effective public participation tool. Perhaps for some municipal planning projects it is effective. My experience with municipal waste management planning leads me to believe that large public meetings are completely ineffective. Worse still, they leave the participating public more angry, frustrated and united in their opposition to the proposed project. That has not stopped the regional municipality that I work for and many others from resorting repeatedly to this format for dissemination of information and public input.

There are a number of other public participation tools at your disposal that are a far more effective means of involving the public and reducing the negative reactions to waste facility siting. Having made that statement, I should also add that my suggestions are more costly and time consuming than the community centre approach.

Plan your public input strategies for small groups of individuals. This will require some advance planning to divide your public into manageable groups. This, however, is time well spent. Identification of issues, special interest groups, particular community needs and attitudes are all valuable tools for municipal waste management planning.

With your publics identified, consider setting up workshops or other small format discussion models with each of those groups. Identification of publics and workshop sessions should begin as soon as the municipality has identified a need for a new waste management facility. Unlike town-hall meetings, workshops are an ongoing process that should be repeated throughout the planning and development stages. If the municipality is seen to be flexible in its willingness to meet with the public, that in itself is a valuable public relations strategy. The municipality and its consultants are seen to be accessible and the public are given numerous opportunities to provide input.

(5) QUALITY OF INFORMATION EXCHANGE

Peelle and Ellis in their 1986 paper stress not only the format of interaction with the public but also the manner in which the information exchange takes place.[1] Municipal waste management planning should be a constant interchange between municipal proponent and impacted public. Every effort should be made to reduce and eliminate the public's frustration in asking questions and getting answers.

Peelle and Ellis state specifically that the public is entitled to direct answers to the specific questions they ask, even where the available data makes that difficult. -If the effect of your answer is to alter the original question or (worse yet) ignore what was really asked then you will face an angry and frustrated public. The public are already highly sensitive to the manner in which they are being treated. Failure to deal directly and candidly with the questions asked may leave people with the impression that you are not listening. In the worst possible scenario, you may be accused of collusion, deceit and abandonment of public duties.

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[1] Op. Cit. Peelle, E. and Ellis, R.

(6) COMMUNITY CONTROL IN FACILITY MANAGEMENT

The NIMBY phenomenon stems from the public's perception that even perfectly designed technologies are no insurance against the human factor entrusted with the responsibility for managing those technologies. With that principle in mind, great gains can be made with the host public through the negotiation of community control mechanisms. The degree and sophistication of those control systems will be a reflection of the particular community involved.

Community control mechanisms may take many forms. Public monitoring groups may be established as soon as the waste management need has been identified. This allows the municipality to endorse a formal mechanism for review and comment of their planning practices, data collection and technology selection.

Any number of public review boards or task forces can be established to monitor construction, plant operation, performance standards, safety measures and emergency response mechanisms. Presumably, these `boards' will be established in response to particular concerns of the host municipality. The effect of the decisions reached by these boards will be a balance between the public feeling they can stop the process when faced with a problem, and plant management's need for autonomy to run an effective operation. Michael Elliott Poirier described it best when he defined effective community control measures as an "interactive decision making structure".[1]

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[1] Op. Cit. Elliott, Michael Poirier.

(7) NON-ADVERSARIAL SITE SELECTION MODEL

Audrey Armour, in her paper entitled, "Is Facility Siting a No-Win Situation?"[1] develops an interesting five-part model for non-adversarial site selection. Her model is based in part on the hazardous waste management facility built by the Province of Alberta in 1980.

In siting the facility, Alberta Environment asked for a volunteer community. The only siting carried out by the province was preliminary constraint mapping designed to eliminate unsuitable areas. With this process completed, information on the identified suitable areas was distributed to all the affected municipalities. Ms. Armour refers to the process as "invitational"[2]. The request for a volunteer immediately eliminates the coercive undertones of most siting projects.

The second step to non-adversarial site selection is to ensure that the community is a partner in the resolution of problems and the making of all decisions. This overlaps with another component of Ms. Armour's model which is to assure the community a right to select among the identified technical options and impact management measures.

Another component of the non-adversarial model requires the implementation of compensation and mitigation measures designed to augment local community benefits. The net result must be to satisfy the host community that the benefits outweigh the costs.

Finally, as a reflection of the growth in environmental sensitivities and consciousness, the non-adversarial model must satisfy the community that safety of the environment and human health are not compromised. No degree of mitigation measures will be satisfactory (even to a volunteer community) if you cannot also convince them that there is no threat to their families' well being and the sanctity of the environment.

Perhaps many would say that the Province of Alberta was fortunate in finding a volunteer community. On a municipal scale, the likelihood of volunteer siting may be dramatically reduced. Without a volunteer, you are back to the current siting mechanisms.

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[1] Op. Cit. Armour, Audrey "Is Facility Siting a No-Win Situation?"

[2] Ibid., at p. 14.

In addition, volunteer communities raise the planning issue as to whether or not you have found the 'best available site'. Certainly for a landfill operation, the characteristics of soils and water have a greater impact than community reaction. You may find yourself having a harder time defending a volunteered site over a technically superior piece of land. With incinerator facilities, fortunately, the physical characteristics of the land are of little significance to the siting of the project.

(8) ENVIRONMENTALLY SENSITIVE POLICIES

This is not a specific recommendation for dealing with a particular public. Rather, it is a suggestion that we begin shifting both our practices and regulatory framework towards more environmentally benign policies. The public has now prioritized environmental concerns and municipal governments have a responsibility to begin reflecting those concerns in both their policies and practices.

Municipalities will gain the respect of their electorate and advance their waste management credibility by implementing wide ranging waste reduction policies. A number of Ontario municipalities have already begun to implement residential recycling programs, household hazardous waste depots, distribution of composting containers and bans on the landfilling of corrugated cardboard. Any measure designed to redirect materials out of the waste stream is an important step to regaining the trust and respect of the community. Non-productive disposal (be it through landfill or incineration) must be seen as the choice of last resort.

CONCLUDING COMMENTS

NIMBY is as complicated and diverse as the people who are its voice. The one quantifiable characteristic of NIMBY is its increasing power and sophistication. The materials set out above were designed to make you think about some of the underlying forces fueling NIMBY. To understand why people have NIMBY reactions and the larger social conscience in which those reactions emerge is essential to change our approach to waste management. Our garbage problems won't go away and neither will NIMBY. Hopefully, somewhere between the two extremes is a livable compromise.

The overriding goal must be to reverse the hostile, polarized siting situation we all live with now. I can make suggestions such as trade-offs, negotiation, mediation, power sharing and compensation - but the `bottom line' is change. Municipalities must change their planning mind-set when dealing with waste management, regulatory authorities must change the legislative framework to reflect the current priorities on the environment and the public must temper their hostilities and distrust and continue to work toward a viable waste management solution.

No site is no solution.

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SITING A RESOURCE RECOVERY FACILITY: COMMUNITY DECISION-MAKING, RISK EDUCATION, AND MULTI-FACETED COMPENSATION

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ABSTRACT

Siting a resource recovery facility has become an emotionally-charged issue in "host" communities where a resource recovery facility is proposed. As a consequence, numerous projects have been delayed or rejected. In an effort to determine the cause(s) for such hesitation or rejection, and identify means to improve siting and community acceptance, the Northeast Regional Biomass Program administered by the Coalition of Northeast Governors (CONEG) Policy Research Center undertook preparation of two studies: one on the environmental and health risks of resource recovery as compared to landfilling, and one on host community compensation practices. Based on the CONEG work and other studies conducted by the authors, the paper will first compare risks of resource recovery and landfill. Then with respect to resource recovery, the paper will review host community compensation practices that have been utilized.

1.0 INTRODUCTION

Siting and acceptance of resource recovery projects, i.e. waste incineration facilities with energy recovery, has become an emotionally-charged issue in communities which would "host" a resource recovery facility. Numerous projects have been delayed or rejected. In each case, one or more of the following circumstances was contributory: (1) the community felt it was shut out of the project planning and siting decision process, (2) the community developed an exaggerated sense of the risks of resource recovery compared with alternative methods of managing solid waste, (3) the community felt it was not being adequately compensated for environmental and other risks of impact on the community, and (4) political activists convinced the community that environmentally-attractive methods such as source-reduction, recycling, and composting can today replace resource recovery and landfill facilities for management of the majority fraction of the solid waste stream.

In an effort to better understand why certain resource recovery projects have been delayed or rejected and to identify potential means to improve the siting process and community acceptance, the Northeast Regional Biomass Program administered by the Coalition of Northeast Governors (CONEG) Policy Research Center recently prepared two studies. In the first study, risks associated with the two solid-waste management "workhorses," resource recovery and landfill, were put on an equal footing for a quantitative comparison. In the second study, host community compensation practices were identified (both legislated and negotiated) including: community participation programs aimed at actively involving the public in project siting and planning; community empowerment; payments in lieu of taxes; monetary payments based on facility use; insurance and trust funds; special services/grants; property value guarantees; and local hiring/procurement policies.

Based on the CONEG work and other studies conducted by the authors, the paper will first compare risks of resource recovery and landfill. Then, with respect to resource recovery, the paper will review host community compensation practices that have been utilized.

2.0 COMPARISON OF HEALTH RISKS BETWEEN RESOURCE RECOVERY FACILITIES AND MUNICIPAL LANDFILLS

For solid waste facilities potential risks include:

- risks to public health;
- risks to public safety; and
- risks of environmental degradation.

The potential risks to public health result from exposure to air and water pollutants released to the environment. Public safety risks are normally associated with the potential for explosions and truck traffic safety hazards. Potential environmental impacts include air and water quality, noise, odor, traffic, aesthetics, litter and dust.

While risks to public safety and the risks of environmental degradation are of great importance in assessing the acceptability of a resource recovery facility, potential risks to public health arising from air pollutant emissions have received most attention and appear to be the largest cause for concern on the part of the public. Presented in this section is a summary of information relating to potential health risks of air emissions from resource recovery facilities. In addition, a comparison is made to potential health risks for municipal landfills, the predominant waste disposal technique utilized in the United States. The comparison is made to illustrate the relative risks of the two most heavily used disposal technologies and to point out that no disposal technology is without risk.

SOURCES OF HEALTH RISK

Air pollutant emissions are the principal source of concern for potential health risks associated with resource recovery facilities, and such risks have been the subject of considerable study. There is also concern over the possible risk of water-supply contamination as a consequence of land disposal of the ash residue from these facilities. Scant quantitative information is publicly available, however, regarding the levels of health risk associated with ashfills. The U.S. EPA plans new regulations under RCRA, Subtitle D, that would require liners to be placed beneath ashfills, and all rainwater that filters through the ash to the liners to be collected, and this leachate tested for unacceptable levels of toxic pollutants prior to discharge. Such control requirements are being implemented to preclude significant risk of groundwater contamination from ashfills.

For municipal waste landfills, the principal source of health risk perceived by the public is risk of water-supply contamination. Indeed, instances of groundwater contamination have occurred with some existing landfills. Little quantitative information regarding such potential risk for landfills of modern design is available, however, at this time. EPA is planning regulations under RCRA, Subtitle D, that would impose design requirements on new municipal waste landfills in order to preclude significant risk of groundwater contamination. Such design requirements are to include liners, leachate collection, and groundwater monitoring.

A demonstrably significant source of health risk at municipal landfills is the emissions from landfills of toxic air pollutants. Although the public is virtually unaware that landfills commonly emit toxic air pollutants, considerable study has been carried out recently to assess the potential health risk from landfill gas emissions.

ASSESSING HEALTH RISK

With air pollutant emissions from both resource recovery and landfill facilities, it is carcinogenic risk, i.e. risk of cancer, that is the primary public health issue. This is because significant carcinogenic risk can result from long-term exposure to even minute pollutant concentrations.

Cancer risk is assessed in terms of an upper-bound estimate of the number of cancer cases to be expected if a population of one million people were exposed to the maximum emissions impact from a facility, 24 hours per day, every day, for a 70-year lifetime. These assumptions help to ensure conservatism in the risk assessment.

Judging the significance of an estimated level of cancer risk is difficult, because this requires subjective judgment regarding what level of risk is acceptable. Most often this judgment is made case-by-case. Based, however, on the historical actions of regulatory and health agencies which perform and review risk assessments, a maximum individual cancer risk of less than 1 case per million people exposed is normally considered "insignificant" risk, and risk levels between 1 to 10 cancer cases per million exposed are considered "acceptable," providing all reasonable means to reduce risk have been adopted. This mirrors the policy(1) on acceptable risk levels set by the South Coast Air Quality Management District in California, and is consistent with policy statements by other states, for example, determinations(2) by the Department of Environmental Conservation in New York State.

HEALTH RISKS FOR AIR POLLUTANT EMISSIONS FROM RESOURCE RECOVERY FACILITIES

Both operating and proposed resource recovery facilities have been the subject of numerous risk assessments since 1985. New York State has led the Country in performing risk assessments on operating plants, based on actual, tested emissions. A summary of risk assessments performed for five energy recovery facilities operating in New York is presented in Table 1.(2) Four of the facilities have ESPs for particulate control, one facility has no particulate control. None of the facilities is scrubber-equipped. Risk was estimated for inhalation exposure to dioxins and furans in each case. The risks at four of five facilities were determined to be in the acceptable range of 1 to 10 in a million. At the fifth facility, the estimated risk of 21 in a million was deemed unacceptable, and the State has required design and operational improvements to reduce dioxin emissions and the associated risk to an acceptable level.(2)

In Table 2, a summary is provided of risk assessments performed since 1985 for 11 resource recovery facilities planned in Pennsylvania, New York, California, and Massachusetts. The planned facilities range in size from 500 TPD to 3000 TPD capacity, and would be located in urban, rural, and mixed landuse settings. All would incorporate scrubbers and high-efficiency ESP's or baghouses. Most of the risk assessments considered exposure to emissions of dioxins, furans, other organic compounds, and trace metals. Most also considered exposure to emitted pollutants by various exposure pathways: inhalation; ingestion and dermal contact with contaminated soils (children are especially exposed to soils); and the food chain. Food chain exposure occurs, for example, by ingestion of reservoir water and vegetables exposed to emitted pollutants, and by ingestion

TABLE 1: ESTIMATED HEALTH RISKS FOR TESTED EMISSIONS FROM ENERGY RECOVERY FACILITIES
IN NEW YORK STATE (a)

Facility	Flue-Gas Setting	Exposure Control	Pollutants Pathways	Carcinogenic Risk(1) Considered	(Chances per Million)
Oneida (200 TPD)	Urban & Rural	ESP	Inhalation	Dioxins, Furans	1
Cattaraugus (112 TPD)	Rural	None	Inhalation	Dioxins, Furans	7
Westchester (2250 TPD)	Rural & Urban	ESP	Inhalation	Dioxins, Furans	2
Albany Answers (RDF-600 TPD)	Urban	ESP	Inhalation	Dioxins, Furans	6
Occidental (RDF-2400 TPD)	Urban	ESP	Inhalation	Dioxins, Furans	21

(a) Upper bound estimate of the number of cancer cases per million people exposed, based on continuous exposure to maximum impacts 24 hours per day, every day, for a 70-year lifetime.

Data Source: Reference 2

TABLE 2: RECENT HEALTH RISK ASSESSMENTS PREPARED FOR PLANNED ENERGY RECOVERY FACILITIES

Facility	Setting	Flue-Gas Controls	Exposure Pathways	Pollutants Considered	Carcinogenic Risk* (Chances per Million)	Data Reference
Lancaster Co., PA (1200 TPD)	Rural	Scrubber, Baghouse	Inhalation, Soil Ingestion, Food Chain, Dermal Contact	Dioxins, Furans Other Organics, Trace Metals	4	4
N. Hempstead, NY (990 TPD)	Urban	Scrubber, Baghouse	Inhalation, Soil Ingestion, Food Chain, Dermal Contact	Dioxins, Furans Other Organics, Trace Metals	1	5
Erie, PA (684-710 TPD)	Urban	Baghouse, Furnace Scrubber	Inhalation	Dioxins, Furans	0.3	6
York Co., PA (1344 TPD)	Rural Industrial	Scrubber, Baghouse	Inhalation, Soil Ingestion, Food Chain, Dermal Contact	Dioxins, Furans, Other Organics, Trace Metals	5	7
Brooklyn NY (3000 I. ,	Urban	Scrubber, Baghouse	Inhalation, Soil Ingestion, Food Chain, Dermal Contact	Dioxins, Furans, Other Organics, Trace Metals	2	8

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* Upper-bound estimate of the number of cancer cases per million people exposed, based on continuous exposure to maximum impacts 24 hours per day, everyday, for a 70-year lifetime.

TABLE 2 (continued): RECENT HEALTH RISK ASSESSMENTS PREPARED FOR PLANNED ENERGY RECOVERY FACILITIES

Facility	Setting	Flue-Gas Controls	Exposure Pathways	Pollutants Considered	Carcinogenic Risk* (Chances per Million)	Data Reference
Washington Co., NY (400 TPD)	Rural	Scrubber, ESP	Inhalation, Soil Ingestion, Food Chain, Dermal Contact	Dioxins, Furans	10	9
Boston, MA (1516 TPD)	Urban	Scrubber, ESP	Inhalation, Soil Ingestion, Dermal Contact	Dioxins, Furans Other Organics, Trace Metals	2	10
Philadelphia, PA (2250 TPD)	Urban	Scrubber, Baghouse	Inhalation, Soil Ingestion, Food Chain, Dermal Contact	Dioxins, Furans Other Organics, Trace Metals	1	11
Stanislaus Co., CA (500 TPD)	Urban & Rural	Scrubber, Baghouse	Inhalation Soil Ingestion, Food Chain, Dermal Contact	Dioxins, Furans Other Organics Trace Metals	2	12
San Marcos, CA (1600 TPD)	Urban	Scrubber, Baghouse	Inhalation	Dioxins, Furans Other Organics,. Trace Metals	3	13
Irwindale, CA (3000 TPD)	Urban	Scrubber, Baghouse	Inhalation, Soil Ingestion, Food Chain, Dermal Contact	Dioxins, Furans Other Organics Trace Metals	5	14

of meats and dairy products from animals that grazed on contaminated forage. Human mother's milk can be a food-chain source of exposure for infants in the same manner.

As shown in Table 2, with the emissions controls planned, risks estimated for the eleven facilities are within the acceptable range of 1 to 10 cases per million. These risk estimates reflect the cumulative risk from all pollutants and all exposure pathways analyzed.

EPA has conducted generic assessments(3) of inhalation-related risks for resource recovery facilities of both older and modern designs. For those existing plants having the poorest combustion efficiency, mediocre control of particulate emissions, and no scrubbers, EPA estimated risks as high as 1000 cases per million. This risk level is well beyond the acceptable range and should prompt more refined assessments to evaluate whether some plants of older design require retrofitting with modern air pollution controls.

With regard to modern plants, however, plants that feature high combustion efficiency, baghouses, and scrubbers, EPA calculated the risk to be in the range of 1 to 10 cases per million. The EPA assessment shows, therefore, that risks associated with modern facilities are within the acceptable range.

HEALTH RISKS FOR AIR POLLUTANT EMISSIONS FROM LANDFILLS

The significant concentrations of carcinogenic pollutants emitted from landfills, and the fact that such emissions seem to be common with municipal landfills, has led a number of both state regulatory agencies as well as the EPA to conclude that landfills operating without gas controls pose a significant risk to public health. These same agencies are now requiring implementation of controls that they believe will bring risk levels to within acceptable levels.

Only a limited number of quantitative risk assessments have been performed to-date for toxic air-pollutant emissions from individual landfill facilities, and these have, for the most part, been "first-cut" assessments. The results of risk assessments performed for two landfills not having gas controls are shown in Table 3. The risk in the

TABLE 3: ESTIMATED HEALTH RISKS DUE TO INHALATION OF TOXIC AIR POLLUTANTS FROM MUNICIPAL LANDFILLS

Landfill Location	Status	Gas Control	Pollutants Considered	Carcinogenic Risk (Chances per Million)	Reference
San Diego, CA (2,250 TPD)	Hypothetical Landfill(a)	None	Benzene, 8 other organics; vinyl chloride not studied	430	17
Montgomery Co., MD (1,800 TPD)	Hypothetical Landfill(a)	None	Vinyl chloride, benzene, other organics	1-50(b)	18,19
Sylmar, CA (unk. TPD)	Active	Flare	Vinyl chloride, other organics	<1	20
Simi Valley, CA (est. 600 TPD)	Active	Engine	Vinyl chloride, other organics	17	21

(a) Risks for a hypothetical landfill were assessed as the "alternative" to a proposed energy-recovery facility.

(b) The range is indicative of analytical uncertainty.

first case was found to be 430 cancer cases per million people exposed, and in the other case to be up to 50 cases per million exposed. Risk assessments performed for two other facilities assumed to have gas controls in place are also summarized in Table 4, with the estimated risk for one facility at 17 cases per million, and for the other, less than one in a million.

EPA has recently completed (15, 16) a generic risk assessment for landfills without gas controls and found the risk to be in the range of 100 to 10,000 cases per million people exposed. This risk is a full order of magnitude greater than has been indicated as a worst-case risk for older, poorly-designed resource recovery plants.

The limited number or risk assessments summarized in Table 3 as well as EPA's generic risk assessment, tend to confirm the position of the regulatory agencies that uncontrolled landfill gas emissions pose a potentially significant risk, if one presumes "acceptable" cancer risk to be in the range of 1-10 cases per million. The limited data also suggest that gas controls have the potential to bring these risks into the acceptable range.

SUMMARY OF COMPARATIVE RISKS

1. This paper has focused on a comparison of health risks associated with emissions of toxic air pollutants from resource recovery versus landfill facilities. While the focus here is on risks stemming from air pollutant emissions, it is recognized that the landfilling of municipal waste, as well as the landfilling of ash residue from energy recovery facilities also present a risk of contamination to groundwater. Little quantitative information has been developed to-date, however, regarding health risk associated with the potential for groundwater contamination by landfills and ashfills. While not addressed in the present study, this issue merits systematic, quantitative investigation, and subsequent public discussion. EPA is presently developing design and operational guidance for landfilling of both solid waste and ash residue in order to preclude significant risk of water contamination.

2. While determination of what constitutes an acceptable level of risk is highly subjective, cancer risk of less

TABLE 4: SURVEY RESULTS: STATE REGULATORY REQUIREMENTS FOR HOST COMMUNITY COMPENSATION FOR RESOURCE RECOVERY FACILITIES

STATE	COMPENSATION TYPE*								REMARKS
	1	2	3	4	5	6	7	8	
Connecticut	None								
Delaware	None								
Maine	X								
Massachusetts			X						
Maryland	None								
New Hampshire	None								
New Jersey				X					
New York	None								
Pennsylvania				X					
Rhode Island			X						
Vermont	None								

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*1	Community Participation and Empowerment
2	Public Service Payments
3	Payments in Lieu of Taxes (PILOT)
4	Monetary Payments Based on Facility Use
5	Insurance and Trust Funds
6	Special Services and Grants
7	Property Value Guarantees
8	Local Hiring and Procurement Policies

than 1 chance in a million is normally considered insignificant, and risk in the range of 1 to 10 in a million is generally considered acceptable, based on past determinations by regulatory agencies.

3. Resource recovery facilities emit quantities of carcinogenic pollutants to the air. Of principal concern are the carcinogens, dioxins, furans, and trace metals. With regard to health risks posed by resource recovery facilities operating today, New York State performed assessments for all operating resource recovery facilities in the State, based on tested emissions. Risks for four of five plants tested were within the range of acceptable risk of 1 to 10 in a million. This is evidence that a majority of operating facilities likely do not pose a significant health risk, and that testing of operating facilities can identify the minority of facilities that requires remedial action.

4. Best Available Control Technology is now well-defined for modern energy recovery facilities (scrubber, with baghouse or ESP), and is implemented on new facilities as a matter of course. An EPA risk assessment determined that for older, less-efficient facilities, not equipped with such modern emissions controls, risk levels can be as high as 1000 cases per million. This is well above the acceptable range of risk, and indicates the advisability of retrofitting some older plants with modern controls. For resource recovery facilities that use modern controls, however, comprehensive risk assessments for numerous such facilities consistently indicate the expected risk level to be within the acceptable risk range of 1 to 10 per million. Air pollutant emissions from planned resource recovery facilities of modern design are normally shown not to pose a significant health threat.

5. Gas produced as waste decomposes in a landfill contains methane, significant quantities of reactive hydrocarbons (VOC), and quantities of toxic pollutants. Methane is also able to migrate beneath the landfill surface where it can enter the basements of homes nearby. Explosions of landfill methane gas have destroyed homes and caused injuries. Emissions of VOC are believed to contribute significantly to existing violations of the ozone ambient standards in the Northeast. Landfill gas contains quantities of toxic pollutants, of principal

interest, the carcinogenic pollutants, vinyl chloride and benzene. Vinyl chloride is potentially produced in landfills during bacterial decomposition of the waste.

6. Most existing landfills (and many planned landfills) do not incorporate any landfill gas controls such as flares or gas-to-energy engines. Vinyl chloride is commonly emitted to the air in significant quantities from these landfills. In some cases, significant levels are also recorded offsite. A number of states have deemed vinyl chloride emission levels to pose a potential health hazard. The limited risk assessments performed to-date for uncontrolled emissions of gas from individual landfill facilities also indicate unacceptable risk levels, approximately 50-400 cases per million. A generic risk assessment performed by EPA indicates risk levels in the range of 100 to 10,000 cases per million is possible for uncontrolled gas emissions from landfills.

7. California, New Jersey, and Wisconsin have recently decided to impose specific requirements for landfill gas testing, as well as requirements for landfill gas collection and control. EPA has also proposed new regulations that would significantly limit emissions from both existing and new landfills, in turn, necessitating landfill gas controls at many landfills nationally.

8. Best Available Control Technology can significantly reduce emissions of landfill gas and the toxic pollutants it contains. Landfill gas collection systems can capture 50%-90% of the gas generated in the landfill, with the remaining fraction escaping to the air as "fugitive" emissions. Control devices such as flares and engines remove up to 95% of the toxic pollutants from the fraction of the gas that is collected. Regulatory agencies believe that such controls will reduce toxic pollutant emissions from landfills, and associated health risks to acceptable levels. Risk assessment studies carried out for two landfills with emissions controls yielded risk levels of less than 1 in a million in one case, and 17 in a million in the other. These results are consistent with the agencies' expectation of significant mitigation.

9. Emissions of toxic pollutants from planned, new resource recovery facilities (and likely the majority of facilities currently in operation) pose acceptable levels of risk, i.e. risks in the range of 1 to 10 in a million. By contrast, emissions of toxic pollutants from municipal landfills without landfill gas controls (i.e. most existing and many planned landfills) pose a potential health risk that exceeds acceptable levels. Implementation of controls on landfill gas emissions has the potential to reduce risks to within acceptable levels; however, this will need to be confirmed by additional field testing and risk assessment of emissions from landfills where controls are installed.

3.0 HOST COMMUNITY COMPENSATION

The types of host community measures for the mitigation of impacts and compensation for the risks of siting resource recovery facilities can be classified into the following eight (8) categories:

1. Community Participation and Empowerment (community participation beyond regulated permit hearings)
2. Public Service Payments (improvements to community infrastructure or public services)
3. Payments in Lieu of Taxes (PILOT)
4. Monetary Payments Based on Facility Use (Toll Agreement)
5. Insurance and Trust Funds
6. Special Services and Grants
7. Property Value Guarantees
8. Local Hiring and Procurement Policies

The measures presented herein can be used to mitigate and/or compensate a host community for a resource recovery facility's impact on the environment, health and safety, public services, finances, property values, and community self-image. The compensation provided does not have to be limited to monetary payments. For example, public service payments may include road, water and sewer system improvements.

The remainder of this paper provides the results of a survey conducted in early 1988 as part of the CONEG study on host community compensation. The survey was focused in the Northeast and included state agencies responsible for solid waste management and host communities and agencies in states with resource recovery facilities in operation, construction, or an advanced planning stage. The resource recovery facilities were selected to represent a cross-section of sizes, developers, and ownership (public or private). Fifty-one host communities with planned or operational resource recovery facilities were contacted.

SUMMARY OF SURVEY RESULTS

Table 4 presents current state regulatory requirements for host community compensation in the Northeast. Five states (Maine, Massachusetts, New Jersey, Pennsylvania and Rhode Island) have regulatory requirements for host community compensation. In Massachusetts, New Jersey, Pennsylvania and Rhode Island, regulatory requirements call for a PILOT (Payment In Lieu Of Taxes) or a monetary payment based on the facility use, i.e. the number of tons of waste processed. A PILOT can be either a lump-sum payment, or a payment based on facility use. Maine requires the establishment of a fund by the developer for the community's use for review of the proposed facility's design, operational specifications, and environmental impacts.

Table 5 describes current practices utilized in the Northeast to compensate host communities. Monetary payment to the host community based on a facility's use is the most prevalent compensation measure. Other frequently used compensation practices are community participation, PILOT payments, provision of special services and grants, and public service payments. Property value guarantee policies were not used in any compensation package for the surveyed facilities.

Table 6 presents information for resource recovery facilities in operation or under construction in California.

TABLE 5: SURVEY RESULTS: CURRENT PRACTICES (ACTUAL/PROPOSED) FOR HOST COMMUNITY COMPENSATION IN THE NORTHEAST FOR RESOURCE RECOVERY FACILITIES (March 1988)

STATE	COMMUNITY	FACILITY NAME	DEVELOPER	SIZE (TONS/DAY)	FACILITY STATUS*	COMPENSATION TYPE** 1	2	3	4	5	6	7	8
CONNECTICUT	BRIDGEPORT	--	WHEELABRATOR(a)	2250	C	(NIR)***							
	HARTFORD	MID-CT	COMBUSTION ENGINEERING	2000	C		X						
	WALLINGFORD	--	VICON	420	C	X	X	X					
DELAWARE	PIGEON POINT	DELAWARE RECLAMATION	RAYTHEON	1000	O (1984)	NONE							
MAINE	AUBURN	--	CONSUMAT	200	O (1981)	X							
	BIDDEFORD	--	--	--	--	(NIR)							
	PENOBSCOT	--	--	--	--	(NIR)							
	PORTLAND	--	DRAVO	500	C	X		X(b)					
MASSACHUSETTS	BOSTON	--	AMERICAN REF-FUEL	1500	P	X	X	X					
	MILLBURY	--	WHEELABRATOR(a)	1500	O (1987)			X			X		
	PITTSFIELD	--	VICON	240	O (1981)			X			X		
	ROCHESTER	SEMASS	ENERGY ANSWERS	1800	C			X			X		
	SPRINGFIELD	--	VICON	360	C	(NIR)		X					

* P = in planning; C = under construction; O = in operation (year initiated commercial operation);

** 1 Community Participation and Empowerment
2 Public Service Payments
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4 Monetary Payments Based on Facility Use
5 Insurance and Trust Funds
6 Special Services and Grants
7 Property Value Guarantees
8 Local Hiring and Procurement Policies

Notes
(a) Formerly Signal Environmental Systems
(b) Ash landfill site also receives PILOT

*** (NIR) = No Information Received

TABLE 5: SURVEY RESULTS: CURRENT PRACTICES (ACTUAL/PROPOSED) FOR HOST COMMUNITY COMPENSATION IN THE NORTHEAST FOR RESOURCE RECOVERY FACILITIES (March 1988)

STATE	COMMUNITY	FACILITY NAME	DEVELOPER	SIZE (TONS/DAY)	FACILITY STATUS	COMPENSATION TYPE 1	2	3	4	5	6	7	8
MARYLAND	BALTIMORE	BRESCO	WHEELABRATOR(1)	2250	O (1985)				X				
	--	MONTGOMERY COUNTY	NOT SELECTED	1800	P		X						
NEW HAMPSHIRE	CLAREMONT	--	WHEELABRATOR(1)	200	O (1987)	NONE							
	CONCORD	--	WHEELABRATOR(1)	500	P	NONE							
	DURHAM	LAMPREY	CONSUMAT	[illegible]	O	(N/A)							
	EXETER	--	WHEELABRATOR(1)	500	P	IN DISCUSSION							
	MANCHESTER	--	VICON	540	P	IN DISCUSSION							
NEW JERSEY	CAMDEN	--	FOSTER-WHEELER	[illegible]	P	(N/A)			X				
	KEARNY	HUDSON COUNTY	OGDEN MARTIN	1500	P				X (Amount in Negotiation)				
	NEWARK	ESSEX COUNTY --	AMERICAN REF-FUEL	2300	P	(N/A)			X				
	OXFORD	WARREN COUNTY	BLOUNT	400	C		X		X		X		
	PASSAIC	-- --	FOSTER-WHEELER	1300	P	(N/A)			X				
	PENNSAUKEN	--	OGDEN	500	P				X		X		
	RAHWAY	--	OGDEN	1500	P	X			X		X		X
	SALEM	--	--	--	--	(N/A)							
	W. DEPTFORD	--	WHEELABRATOR(1)	575	C	(Negotiations not started)			X				
	WOODBINE	--	NOT SELECTED	500	P				X		X		
NEW YORK	ALBANY	--	ENERGY ANSWERS	600	O (1981)	NONE							
	BABYLON	--	OGDEN MARTIN	750	C	NONE							

TABLE 5: SURVEY RESULTS: CURRENT PRACTICES (ACTUAL/PROPOSED) FOR HOST COMMUNITY COMPENSATION IN THE NORTHEAST FOR RESOURCE RECOVERY FACILITIES (March 1988)

STATE	COMMUNITY	FACILITY NAME	DEVELOPER	SIZE (TONS/DAY)	FACILITY STATUS	COMPENSATION TYPE 1	2	3	4	5	6	7	8
	CATTARAUGUS	--	ENERCON	100	O (1983)	NONE							
	DUTCHESS CNTY	--	--	--	--	(N/R)							
	GLEN COVE	--	(N/R)	75	C	(N/R)							
	MERRICK	HEMPSTEAD	AMERICAN REF-FUEL	2250	C	X							
	HUDSON FALLS	--	FOSTER-WHEELER	400	C				X				
	ISLIP	--	WESTINGHOUSE	510	C	NONE							
	NEW YORK CITY	BROOKLYN	WHEELABRATOR(1)	3000	C	X							
		BRONX	NOT SELECTED	2000	P	X							
		MANHATTAN	NOT SELECTED	2000	P	X							
		QUEENS	NOT SELECTED	2000	P	X							
		STATEN ISLAND	NOT SELECTED	3000	P	X							
	OSWEGO	--	CONSUMAT	200	O (1985)	NONE							
	OYSTER BAY	--	--	--	--	(N/R)							
	PEEKSKILL	WESTCHESTER COUNTY	WHEELABRATOR(1)	2250	O (1984)		X		X		X		
	--	WASHINGTON COUNTY	--	--	--	(N/R)							
PENNSYLVANIA	E. STROUDSBURG	MONROE COUNTY	NOT SELECTED	300	P	X	X		X	X			
	BETHLEHEM	LEHIGH COUNTY	AMERICAN REF-FUEL	1000	P	X			X				
	[illegible]	BLUE MT.	OGDEN	1000	P	X			X				
	ERIE	--	CONVERSION	850	P			X					
	MANCHESTER TWP	YORK COUNTY	WESTINGHOUSE	1300	C		X		X		X		
	PLYMOUTH TWP	MONTGOMERY COUNTY	[illegible]	1200	P	(NEGOTIATIONS NOT STARTED)							
RHODE ISLAND	N. KINGSTOWN	QUONSET POINT	SIGNAL	750	P	X		X			X		
VERMONT	RUTLAND		VICON	240	O (1987)	NONE							

TABLE 6: SURVEY RESULTS: CURRENT PRACTICES (ACTUAL/PROPOSED) FOR HOST COMMUNITY COMPENSATION OUTSIDE THE NORTHEAST FOR RESOURCE RECOVERY FACILITIES (March 1988)

STATE	COMMUNITY	FACILITY NAME	DEVELOPER	SIZE (TONS/DAY)	FACILITY STATUS*	COMPENSATION TYPE** 1	2	3	4	5	6	7	8
California	Modesto	--	Ogden Martin		C	X			X		X		
	Commerce City	--	Foster Wheeler	350	O (1981)	NONE, CITY PART-OWNER OF FACILITY							

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5 Insurance and Trust Funds
6 Special Services and Grants
7 Property Value Guarantees
8 Local Hiring and Procurement Policies

DETAILED DESCRIPTION OF COMPENSATION PRACTICES

Community Participation and Empowerment

Community participation and empowerment has been limited. To encourage community participation, Maine has established a $50,000 fee (to be paid by the developer) for the community's use to review the project. In New York City, where several resource recovery facilities are planned, the City provides funding to a Citizens Advisory Committee (CAC). The CAC hires its own consultant(s) to assist in the technical and environmental review process (during the preparation of Environmental Impact Statements and Permits) for the proposed facilities. Host communities for five other facilities, East Stroudsburg, Pennsylvania; Rahway, New Jersey; North Kingstown, Rhode Island, Boston, Massachusetts and Wallingford, Connecticut had input on restricting truck access routes through the community.

The Rahway, New Jersey agreement allows for facility access for city and county health officials and city examination of facility books and records. The agreement prohibits process discharge and outside storage of waste (to prevent leachate discharge) to the nearby river. The agreement also requires the building to be aesthetically attractive and present a non-industrial image.

The Boston, Massachusetts, host community agreement is incorporated into that proposed facility's Site Assignment, which is a local permit in Massachusetts issued by a host municipality's Department of Health. The Boston Site Assignment requires semi-annual stack testing by an independent engineer, continuous emissions monitoring with health department access to the monitoring data, automatic plant shut down if emissions exceed limits more than once a quarter, facility design consideration to discourage vector infestation, and spare part and operation/maintenance log keeping requirements.

Interveners (a local consumer and environmental group) will have direct access to the Hempstead, New York, facility emission data records. This facility, through negotiations, will install a baghouse (in lieu of the originally planned electrostatic precipitator) and will have an allowable particulate emission rate that is lower than required by regulation.

No operating facility in the Northeast (as identified in the survey) is being actively and continuously monitored directly by the community or their representatives for emissions. This is being done, however, at a facility in Modesto, California and was stipulated in the Boston Site Assignment.

Public Service Payments

The siting of a resource recovery facility may increase demands on the host community's roads, road maintenance, water supply systems, sewage treatment facilities, or other public services. To compensate for these impacts, public service payments may be negotiated. These payments may be either monetary payments or non-monetary. Improvements to the community infrastructure made by the developer beyond what is needed for the facility itself can be considered improvements made for the benefit of the host community. Examples of such improvements have included:

- Construction of new roads to provide access to the facility that will minimize truck traffic on existing roads;
- Upgrading of sewer lines in the vicinity of the facility site to accommodate not only the facility, but other potential growth in the community; and
- Provision of funding by the developer to construct additional water storage capacity for the community.

Monetary payments based on facility use may also be used to maintain or upgrade public services impacted by the facility.

Payments in Lieu of Taxes (PILOT)

PILOT payments (which may be either annual lump sum payments or payments based on facility use) can be used to compensate the host community for the loss of the property from the community's tax base.

The survey found that most PILOT payments (either required by law or negotiated) are monetary payments based on facility use. Payments range from $1.00 to $2.00 per ton of waste processed.

Two facilities (Westchester County and Portland, Maine) make annual lump-sum PILOT payments. In Westchester County, payments are made to the host community in an amount equal to the taxes that would be due if the property were not tax-exempt, and in Erie, Pennsylvania an amount equal to 25% of the taxes due will be paid.

Monetary Payments Based on Facility Use

Monetary compensation to a host community based on facility use is perhaps the most wide-spread compensation measure. Most of the monetary payments based on facility use are in the $1-$2/ton range with some having escalators.

Insurance and Trust Funds

The only case among survey participants of host community compensation in the area of insurance and trust funds was E. Stroudsburg, Pennsylvania. E. Stroudsburg, through negotiations, was established as a co-insured party with the Monroe County General Authority (the project developer) and was to be provided first dollar coverage for defense of E. Stroudsburg for any action against E. Stroudsburg arising from the siting or operation of the facility.

Special Services and Grants

These measures compensate the host community for impacts related to the community's quality of life, or other impacts of special concern. They can also be considered as incentives to the community to host the facility. Of the facilities surveyed, the most common special service granted to a host community was a modified (reduced) disposal fee.

Property Value Guarantees

Although considered for siting hazardous waste facilities and municipal landfills, regulated or negotiated property value guarantees are a relatively recent area for compensation consideration for resource recovery facilities. Of the facilities surveyed, property value guarantees have not yet been provided. The effect of siting a resource recovery facility on adjacent properties is difficult to assess - facilities may have a neutral or positive effect on residential or industrial property values.

Local Hiring and Procurement Policies

The only negotiated local hiring and procurement policy found among the survey participants was in the agreement between Union County and Rahway, New Jersey. In that agreement, a plan for hiring and staffing (construction and operations) is to be prepared by the selected vendor which will set forth the procedures by which the vendor will, to the extent permitted by the law, grant preference to qualified residents. The vendor is also required to purchase materials from local businesses, whenever possible or practicable.

Notice

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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NEGOTIATION IN MUNICIPAL SOLID WASTE SITING - IT CAN WORK

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ABSTRACT

The siting of municipal solid waste facilities generates controversy. NIMBY (not in my backyard) characterizes the response of citizens to local proposals made by proponents of waste management projects. Proponents tend to focus on the technical feasibility of the operation.

The environmental hearing is the technique commonly applied to the resolution of such disputes. The process is often justified on the grounds that it includes protection of the public interest.

The environmental hearing has its problems, though. It can unnecessarily polarize participants and become excessively expensive due to evidentiary requirements. Hearings are often perceived to degenerate into a forum where lawyers are asserting their self interest, not that of their clients.

Environmental negotiation/mediation is an appropriate alternative to the adversarial hearing. Ideally, the parties themselves will exchange information and concerns more openly and effeciently and may well settle an approach to some or all of the issues. The substance of such settlements may then be contained in a formal contractual agreement, enforceable through the judicial process and/or be embodied in conditions of approval.

The author of this paper has had extensive experience with the siting of municipal, hazardous, liquid industrial and radioactive waste treatment and disposal facilities, acting for both proponents and opponents. Some siting exercizes were successful; others were not. Some triggered lengthy and adversarial hearings; others were dealt with by agreement.

The purpose of this paper is to provide assurance that waste management facilities can be successfully sited and that the siting process does not inevitably result in protracted hearings and court proceedings.

The author will discuss the pros and cons of hearings and the negotiation process and what has and has not worked. He will reflect on the factors which make each situation unique and challenging and the differences which preclude waste disposal conflict resolution by a formula method.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

I. INTRODUCTION

Groundwater. Leachate. Incinerated gases. Dioxin. These are the specific concerns repeated over and over again by citizens facing the possibility of the siting of a municipal solid waste facility in their area. Generally, the overwhelming emotional response is NOT IN MY BACK YARD (NIMBY).

These things are controversial. The greatest controversy may be the mismatch of issues between the citizens and the proponent. Instead of the emotional response of NIMBY and the often unwarranted preoccupation with toxic fallout, proponents focus on the technical feasibility of the proposal behind the facade of rationality.

Hearings are one forum used to resolve conflict. The proponent is forced to match issues with the concerns of ratepayers and the constructive criticisms of commenting government agencies.

A preferable method of dispute resolution is the negotiated settlement. There are many types of negotiated settlement, from the effective public participation program leading to an uncontested hearing, to the involvement from the outset of a third party mediator facilitating negotiation and settlement between all of the interested parties.

II. THE ENVIRONMENTAL HEARING PROCESS

A. REVIEW OF THE PROCESS

The first party to be heard is the proponent. The proponent is the local government or the private company proposing to build the incinerator, the sanitary landfill, the recycling or composting plant. The proponent defines the need for the facility, creates the designs and searches for the best location. It is the proponent's task to convince the other parties, and more importantly the hearing officers, that the homework has been done and the conclusions reached make sense.

The opponents can be government or private corporate competitors, area residents, or other government or non-government organizations, with a direct pecuniary or proprietary interest in the outcome of the decision-making process.

There may also be intervenors or amicus curiae (friends of the court). These are environmental or other advocacy groups, interested individuals or organizations, including neighbouring municipalities or other government agencies. Their participation is usually focused on a single concern or a group of related issues. Often, they have to overcome the legal argument that they have no "standing" to participate in the hearing. Occasionally, they are given full party status. Usually, they can make submissions but cannot cross-examine on the evidence of other parties nor bring procedural motions throughout the hearing.

The cadillac version of the hearing process in Ontario occurs pursuant to the Environmental Assessment Act or the Consolidated Hearings Act (where there are planning considerations or other issues such as expropriation). This process really begins with the presubmission consultation (PSC) before the environmental assessment (environmental impact statement) is formally submitted for government review and comment. At this early stage, the proponent and government review agencies work together to structure the document to address all of the anticipated concerns and issues.

After the formal submission of the environmental assessment, a public comment period is provided, followed by or contemporaneous with the determination of the parties, application for intervenor funding, and preliminary hearings to scope the issues. Witness statements and interrogatories are exchanged, testimony and exhibits are prepared, and the hearing is scheduled. At the preliminary hearing stage and/or at the outset of the hearing, procedural rules are established and motions are entertained.

The proponent presents its case first, followed by the case for the opponents and submissions by intervenors. Each witness or panel of witnesses giving evidence on the same topic is submitted to cross-examination by the other parties to challenge credibility, veracity of the evidence, and to qualify expertsopinions. The hearing officers often have questions of the witnesses as well. In fact the tribunal may appoint its own experts to assist them.

Once all of the evidence has been given, counsel for each party is permitted to summarize the evidence and make legal arguments.

The hearing officers give oral or written reasons for their decisions, or provide a report with recommendations to the decision maker.

B. ADVANTAGES IN THE CONTEXT OF MUNICIPAL SOLID WASTE MANAGEMENT

Hearings are a chance for anyone with a valuable point of view or interest to protect to make submissions, test the veracity of the evidence of other parties, and participate in the decision-making process. The tribunal or other decision-making body usually comes with a reputation for being fair and impartial, rational and unbiased, competent and concerned. The hearing officers who will make the decisions or recommendations are recognized by the parties as having a mandate to protect the "public interest".

Other advantages of a hearing are a product of the first point. The hearing process often legitimizes decisions that have been made. People tend to be happy with decisions if they feel that they have had their day in court and made their points of view known. This is true even if the decision is not the one they wanted.

The hearing setting permits the parties to submit their conflict of fundamental values for resolution by an impartial third party. If the proponents and opponents feel that they have had a fair hearing and their concerns have been understood and taken into account, the hearing can act as a vehicle to defuse hostility and tensions.

All participants given party status can be effective at a hearing despite their relative lack of bargaining power outside of the hearing.

A final advantage of the hearing process is that examination and cross-examination under oath is an excellent way to uncover faulty assumptions or conclusions of experts.

C. DISADVANTAGES IN THE CONTEXT OF MUNICIPAL SOLID WASTE MANAGEMENT

The public interest usually requires the "best" decision. Best means what technological and financial resources will allow. Best means what is safest for the environment and the health of the community, but it can rarely mean a decision which satisfies every concern and need of each of the parties. Someone wins, someone loses.

Because of this, the parties sometimes end up more polarized in their positions then they were before the hearings. This is particularly harmful in environmental disputes where the decision of a hearing is often the prelude to a long term releationship between the parties.

Hearings are expensive and time consuming. Major issues must be dealt with extensively and effectively by the proponent in order to gain approval for the project. The little but important issues may be overlooked. Unfortunately, hearings become a forum for the lawyers and the consultants, rather than for the parties.

D. IMPROVING THE HEARING PROCESS

The subject matter of an inquiry doesn't always have to be dealt with comprehensivly and completely in one big block of time. Conducting a hearing in phases should always be considered as an alternative where a long hearing has many issues, from the broadest planning concepts to the most detailed technological concerns.

Too little attention is paid to the crucial matter of the timing of the hearing. If parties are in fundamental disagreement, where there is a genuine "not in my backyard" approach, the hearing should occur sooner rather then later. If necessary, the hearing could be phased to deal with the fundamental issues first. If and when the rationale and need have been established and the approval of the project seems inevitable, the parties can negotiate conditions of approval, site design, and other mitigating factors. A joint submission can be made on these issues, or where negotiations fail to reach agreement, each side can put forward its proposal for the conditions of approval.

On the other hand, parties which are not at odds over basic value differences can afford to have the hearing at a later date. They can be forced to narrow the issues before the hearing starts and conduct several rounds of negotiations to find solutions to the outstanding issues. This approach precludes certain techniques used by lawyers such as the element of surprise and the "shot gun approach".

To an extent, presubmission consultation, the exchange of witness statements and interrogatories, and the preliminary hearing help to avoid the legalistic traditions which can prolong hearings.

Hearings can be run effectively. There are many cases in point in recent years.

III. ENVIRONMENTAL MEDIATION/NEGOTIATION

Environmental mediation/negotiation is an appropriate alternative to the adversarial hearing. Idealy, the parties themselves will exchange information and concerns more openly and efficiently and may well settle an approach to some or all of the issues. The substance of such settlements may then be contained in the formal contractual agreement enforceable through the judicial process and/or enbodied in conditions of approval.

A. REVIEW OF THE PROCESS

The process is consensus-building to develop an agreed settlement of the issues. Participation in the process should be voluntary. Part of the process is to agree upon the process itself. There are no cut and dry rules as to what must take place.

Mediation is negotiation using an independent third party. The mediator serves at the pleasure of the parties and must have their confidence to proceed.

Once the process has been agreed upon, it must be committed to writing. The mediator must be identified in the agreement, and given some discretionary powers to excercise control in the discussions. Also the agreement must provide for payment of the mediator. In particular, he must be able to convene meetings, propose studies, meet separately with parties, and adjourn the proceedings from time to time when he feels its appropriate.

Another aspect which must be reduced to writing is a list of interested parties and a list of the issues that they are fighting about. If they cannot produce this list, mediation may be premature.

A time limit for the process may be imposed by the agreement. There should be a procedure for having meetings open to the public and to the press, and policies spelled out in advance as to which topics are not to be discussed with the press.

The agreement must provide for the exchange of data and expertise, and the discussions must be on a "without prejudice" basis.

Ultimately, the process is about sharing power. Proponents must learn how to share some decision-making and intervenors, in turn, must abandon the strategy of fight and delay without compromise.

Sharing of decision-making is preceived by most proponents and their lawyers as being an unlawful and unacceptable abdication of authority and responsibility. Municipal politicians and directors of boards are answerable to a constituency which is entitled to hold them responsible.

The problem of power sharing is exaserbated by the trend to large regional waste disposal facilities. Those interested in or directly affected are usually a small minority within the overall constituency of the proponent.

To ensure continuity in the negotiations, each party must appoint representatives who will be able to attend throughout the process. Representatives of less sophisticated intervenors may require pre-mediation coaching to "bring them up to speed" with the negotiation skills of the other participants. This is especially true with the siting of municipal solid waste management facilities.

B. AN APPROACH TO ENVIRONMENTAL MEDIATION IN THE SITING OF MUNICIPAL SOLID WASTE FACILITIES

Conventional wisdom and practice dictate that comprehensive public participation is essential to success for proponents. Reliance on the formal hearing or court system for timely authorization of any given undertaking is risky, costly, and often futile. However, conventional mediation may be seen by the proponents as abdicating too much authority and power over decision-making. The use of a referee from the outset of the process may be the answer.

In contrast to a strict mediator or arbitrator, limits cannot be placed on the period of the referee's involvement. The referee is like a public participation co-ordinator, becoming involved at the earliest stage of planning and ending his involvement when an agreement is reached.

The beginning of any waste management or facility siting process requires the proponent to define the problem or issue. This is a first opportunity for community involvement. The community should be invited to react or respond to the problem/issue as initially defined.

Community opinion must be solicited from a representative cross section of perceptions of the problem/issue. This preliminary exploration phase is iterative, as the proponent checks and modifies his definition for completeness as the community reacts to the evolving definition.

People residing or working within or close to the study area should be represented. Those with direct economic or environmental interests must also be involved. Once the parties are established, the agreement on how to proceed must be committed to writing.

Defining the interests of all the parties helps all the participants. It stimulates an awareness for the context in which the concerns of each party arise.

The proponent must keep an open mind. All parties must be willing to share their perceptions about the problem/issue, as well as identify their legal responsibilities and opportunities. In this manner, the process will remain structured to address the issues and to determine the level of detail and at what time in the planning process the issues are addressed.

When the parties appreciate the various perceptions and are aware of the formally imposed decision-making process, they are encouraged to deal with one another on a more equal basis. Potentially threatening situations are alleviated when the parties are aware why each is involved and what their legal rights are. An understanding is achieved as to what is happening and how the process can be influenced.

Encouraging the parties to offer alternatives to solve the problem creates a sense of shared power. There should be an agreement to consider any proposal, although each party must be honest about what it feels are viable solutions. If the proponent is going to maintain the trust of the interested community parties and have them subscribe to the process and its resulting actions, their suggestions have to be reviewed with sincerity and respect.

The parties must be actively involved in reviewing and evaluating the options. Reasons why a particular alternative will or will not work must be discussed. In developing such a comparative evaluation, the various parties involved will have a more thorough understanding of the most viable solution. In developing this solution, the parties will feel some responsiblity for its implementation.

This sense of responsibility will carry over into the refinement of preferred alternatives. The various parties will be more willing to consider modifications to their initial suggestions once they have agreed on some basic constraints and exclusions. They will act in concert to ensure their basic demands as a group are met.

In refining the alternatives and developing the ultimate recommendation, each party represented must be given responsibility in subsequently enacting the solution. Not only must they be involved, they must be seen to be active in the application of the workable solution. This further binds the parties to the solution, and maintains their credibility in the eyes of their respective constituents and the community as a whole.

Community partnership with the proponent is confirmed when each party's continued responsibilities are acknowledged as the recommended solution is offered. At the same time, community anxiety about how the solution is to work once implemented and how future problems might be addressed is eased.

In addition to being the neutral party respected by all, the referee may be given the authority to make rulings on aspects of the relationship between the parties, to comment formally and publically on the behavior of any party, and to make recommendations.

During a long-term involvement, the referee may lose his perceived neuturality, especially if a series of rulings are made which consistently offend one of the parties. In the final stages of planning and application for approvals, a more conventional mediation process may be invoked. This would likely require a different independant third party.

IV. CONCLUDING COMMENTS

Notwithstanding the risks, the potential benefits to the solid waste facility siting process of removing some of the psychological and fundamental principal objections to power sharing are obvious and significant. The traditional hearing process is a crap shoot. The outcome can never be pre-determined, and ultimately, the decision making is out of the hands of the proponent. By allowing the public to buy into the decision-making process, the proponent retains some of the decision making authority to the very end.

In addition, the process throughout remains focused and constructive when a mediator or referee is used from the outset. Because the community is part of the process, there should be no lasting resentments or doubts that the process has failed any one party.

FORMATION MECHANISMS OF CHLORINATED ORGANICS AND IMPACTS OF SORBENT INJECTION

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ABSTRACT

The formation mechanisms of polychlorinated dibenzo-p-dioxin (PCDD) and polychlorinated dibenzofuran (PCDF) and their precursors were examined in a laboratory reactor. Both homogeneous and heterogeneous reactions were studied between 200 and 800°C with HCl, Cl_2, and phenol as reactants in a simulated flue gas. Analysis of the reactor effluent showed that homogeneous phase production of chlorophenols, potential precursors to PCDD and PCDF, was related to HCl concentration, reaching a maximum formation level around 650°C. However, Cl_2 produced a greater variety of chlorinated aromatics at levels over three orders of magnitude greater than with HCl, with product concentrations reaching maximum formation levels around 350°C. Heterogeneous tests at 450°C using a CuCl catalyst increased formation of chlorinated organics and PCDDs and identified the major chlorinating reactant to be Cl_2. Sorbent injection is proposed as a method of removing the chlorination sources prior to temperature regions of optimal chlorinated organic formation.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for presentation and publication.

INTRODUCTION

Successful control technologies for the minimization or elimination of the formation of polychlorinated organics in waste incinerators depend upon an adequate understanding of the mechanisms of their formation. The most notorious group of these chlorinated organics, noted for their extreme toxicity (1), includes the polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). Current strategies for reducing PCDD and PCDF formation rely upon controlling combustion conditions to promote the complete oxidation of precursor structures (2) or removing the chlorinated organics by condensation and absorption upon the surface of a sorbent, injected as a slurry in a spray dryer application (3). Although the efficacy of the former method is subject to some debate (4), limited tests have shown some success (5). Nonetheless, it is not entirely clear that this method can consistently ensure adequate prevention. Spray dryer results have shown PCDD/PCDF removals of 95%, with the added benefit of simultaneous acid gas control (3). However, this method results simply in media transfer of the toxics from the air stream to the spent sorbent stream by a condensation/ absorption mechanism (3), likely presenting additional disposal problems. It is clear that further understanding of the mechanisms of PCDD/PCDF formation in waste incinerators are crucial toward development of a successful control technology.

Work done in the past on the formation of PCDDs and PCDFs has focused on two major mechanisms, high temperature gas-phase formation and low temperature heterogeneous formation between condensed reactants on fly ash particles. In the former, Eklund and coworkers (6,7,8) found that it is sufficient to combine separate carbon sources as varied as phenol, propane, and methane with oxygen and HCl at 400 to 950°C to synthesize chlorinated compounds. These gas-phase reactions have been observed in sealed reactor tubes (6), sampled gas streams (7), and a tubular reactor (8). These experiments produced a large variety of chlorinated alkanes, -alkenes, -aromatics, PCDDs, and PCDFs, showing that formation of these toxic chlorinated organics can be synthesized from relatively ubiquitous precursors in the combustion gases. This was supported by work with coal decomposition (9) in which PCDDs were produced through chlorination by Cl_2 and HCl gases at 600°C (10 s gas residence time) in a quartz tube furnace reactor.

The existence of a gas-phase formation mechanism has its detractors. Shaub and Tsang (10) used a homogeneous reaction model with detailed equilibrium and chemical kinetic considerations to show that gas-phase formation (>~930°C) of PCDDs is improbable and unable to account for emission levels sometimes found in actual practice. Accordingly, the same driving forces which destroy PCDDs will also lead to decomposition of their precursors, such as chlorophenols.

Shaub and Tsang (11) further noted that the production of PCDDs in laboratory gas-phase experiments may be attributed to hydroxyl groups (OH^-) on the surface of the quartz and glass vessels acting as a reactant source promoting formation. They cited Buser and Bosshardt's (12) work in which conversion of polychlorinated biphenyl (PCB) to PCDF increased for sealed quartz tube experiments when the ratio of tube surface area to reactant quantity increased. Since fly ash tends to be both basic and high in silicon dioxide (SiO_2) content (55%) like the quartz tube surface, Shaub and Tsang suggested that both fly ash and quartz will act similarly to form PCDFs through surface reactions.

The involvement of gas-phase/fly ash particle reactions in production of PCDDs/PCDFs has been further supported by Dickson and Karasek (13), Rghei with Eiceman (14,15,16,17) and Vogg and Steiglitz with Metzger (18,19,20) through detection of PCDD/PCDF on fly ash.

Eiceman and Rghei (15) treated fly ash samples from a municipal incinerator at 200 to 300°C and found heterogeneous reactions with a chlorine source, producing chlorinated organics. The chlorine source used was Cl_2, although HCl was expected to result in significant chlorination, albeit through a different mechanism and rate. These reactions did not appear to result from simple thermal rearrangement of compounds already present on the fly ash.

Dickson and Karasek (13) supported the view of a heterogeneous, surface-catalyzed formation mechanism, demonstrating the formation of PCDD from chlorinated phenols in fly ash.

Heterogeneous formation by reaction of precursors condensed on fly ash was suggested by Vogg et al. (18). Their results showed that chlorinated organics, including PCDDs and PCDFs, could be formed by heating municipal waste ash samples to 300°C. They interpreted this as a secondary mechanism of formation in a low temperature region with primary formation in the combustion zone from waste chlorinated organics. As first proposed by Griffin (21), they postulated that HCl from the fly ash reacted with O_2 via the Deacon process (shown below) to form gaseous Cl_2 which, in turn, reacted with aromatic ring structures, producing chlorinated aromatics:

$$2HCl + 1/2\ O_2 \longleftrightarrow Cl_2 + H_2O$$

The Deacon process was found to be enhanced by the presence of metal choride catalysts, most noticeably $CuCl_2$, in the ash.

The possibility of multiphase synthesis of chlorinated organics from inorganic sources of chlorine was cited as a possiblity by Shaub and Tsang (10). Multiphase production of HCl [and sodium silicate (Na_2OSiO_2)] from silicon dioxide (SiO_2) and sodium chloride (NaCl) was noted (22) in the presence of water (H_2O), while Cl_2 was noted in the presence of oxygen (O_2). The production of HCl was verifed for experiments using refuse (23) and electrostatic precipitator ash (24) with NaCl and SiO_2/Al_2O_3.

This work examines the formation of chlorinated organics in the temperature region from 200 to 800°C, considering both gas-phase and heterogeneous reactions. Both HCl and Cl_2 are evaluated as the chlorine source, while phenol is used as the carbon source. The catalytic role of chlorinated copper compounds and SiO_2 surfaces in promoting the formation of chlorinated aromatics was studied. As such, this work examines the formation of chlorinated organics through both homogeneous and heterogeneous mechanisms, as described above. The role of sorbent injection in thwarting these mechanisms is examined. It is the purpose of this endeavor to elucidate the mechanisms of PCDD and PCDF formation in municipal solid waste incineration such that proper control measures can be taken to reduce or eliminate the formation of these toxics.

EXPERIMENTAL

Experiments were run in a concentric tube, quartz flow reactor inserted into a horizontally mounted, single zone, electric furnace (Figure 1) termed the chlorinated organic reactor (COR). Reactor temperatures were varied from 200 to 800°C. Process gas simulating flue gas composition was preheated by flowing through the outer quartz annulus into the inner tube reaction zone. The total operating flow rate of 0.93 l/min resulted in a reactor residence time of 10 s. The basic process gas consisted of 1% oxygen (O_2) and the balance nitrogen (N_2). The test gases included phenol ($C_6H_{11}OH$), hydrogen chloride (HCl), and chlorine (Cl_2).

HOMOGENEOUS TESTS

Phenol at nominal 1000 ppm was fed into the entrance of the reaction zone by flowing a N_2 carrier gas through a heated tube filled with glass beads coated with liquid phenol. A mass balance determined the actual phenol delivery rate. The reactor exit gases were captured in two methylene chloride ($C_2H_2Cl_2$) solvent impingers in series, located in an ice bath to prevent evaporation. An activated charcoal cartridge followed the impingers to retain any products that may have been missed by the $C_2H_2Cl_2$. The duration of each run was 1 h. All procedures were carried out under a ventilated hood.

At the end of each run, the products and $C_2H_2Cl_2$ in the impingers were transferred to a 500 ml round bottom flask along with the washings from the impingers, quartz reactor, and product lines. The excess solvent was then evaporated with a Snyder column and heating mantle, concentrating the products to ~10 ml. The products were then transferred to a smaller vessel and concentrated to below 1 ml by evaporating under flowing N_2. The walls of the vessel were then rinsed with $C_2H_2Cl_2$ and the sample concentrated to 1 ml for analysis by a gas chromatograph (GC) with a flame ionization detector (FID) and a GC with a mass spectrometer (MS). Further concentration of the GC/MS sample was sometimes necessary to ensure specie detection.

To verify product formation, concentrated COR samples were run on a Hewlett Packard 5890 GC equipped with an FID. A 30 m long by 0.53 mm inside diameter (ID) fused silica DB-5 column was used. Samples were injected at an initial temperature of 60°C held there for 5 min, and then the temperature was increased at 2°C/min. When the temperature reached 90°C, the rate was changed to 8°C/min until obtaining a final temperature of 250°C.

Following GC/FID, products from selected runs were identified by GC/MS analysis on a Hewlett Packard 5988 GC/MS with a 30 m long by 0.25 mm ID fused silica column. Run parameters were the same as with the GC/FID.

HETEROGENEOUS TESTS

These tests were done with some modifications to the homogeneous test procedures. A quartz wool plug was inserted into the middle zone of the quartz reactor's inner tube. The catalyst to be tested, CuCl, was added in 1 g quantities, followed by a second quartz plug. All reagents used in the reactor were extracted with methylene chloride prior to use. With the plug in place, the reactor was run as in the homogeneous tests. At the completion of each run, the quartz plug, reacted CuCl, and rinsings from the impingers, reactor, and product lines were extracted in a Soxhlet apparatus with $C_2H_2Cl_2$ for at least 4 h. The extract was then concentrated in the manner discussed previously.

Cl_2 TESTS

Testing for Cl_2 emissions was done by flowing the reactor effluent through an impinger train filled with potassium iodide (KI) solution. The Cl_2 gas reacts to form iodine (I_2) which can be quantitated by titration with standardized sodium thiosulfate ($Na_2S_2O_3$). These KI tests were done at all run conditions during heterogeneous testing to detect Cl_2.

STDR METHODS

Sorbent reactivity tests with calcium hydroxide [$Ca(OH)_2$] were run in a fixed bed, short time, differential reactor (STDR) (Figure 2) equipped with an air-actuated slider assembly for short residence time exposures. Experiments were done using 1000 ppm HCl and Cl_2 at 400 and 800°C with a commercially available $Ca(OH)_2$. A quartz wool bed was put into the sample port of the STDR with approximately 4 mg of sample dispersed evenly about the bed. The samples were exposed at times ranging from 0.3 to 5 s. After exposure, the quartz wool plug and sample were transferred to a beaker containing 25 ml of eluent [0.005 M sodium carbonate (Na_2CO_3), 0.005 M sodium bicarbonate ($NaHCO_3$)], and 2 drops of concentrated sulfuric acid (H_2SO_4). The sample was stirred for 5 min and filtered into a 50 ml volumetric flask for Ca^{++} analysis by atomic absorption spectrophotometry (AA) and Cl^- analysis by ion chromatography (IC).

X-RAY DIFFRACTION

X-ray diffraction (XRD) analyses for compound identification were performed on a Siemens model D-500 diffractometer with a copper target source running at 50 kV and 40 mA. The entrance aperture was 1.0 degrees and the detector slit was 0.05 degrees. A scintillation detector with a graphite monochrometer was used. Spectra were identified by computer comparison with the Joint Committee for Powder Diffraction Spectra (JCPDS) spectral files.

RESULTS AND DISCUSSION

Homogeneous Phase Formation

The COR was used to examine the effect of temperature and HCl concentration on the formation of chlorinated aromatics. With the basic process gases and phenol, species production varied with changing temperature (200 to 800°C) and HCl concentration (200 to 3300 ppm). Figure 3 shows a representative GC trace from a COR run, indicating the major species formed. Figure 4 demonstrates the temperature effect on the concentration of the major chlorinated product, 4-chlorophenol, at 3000 ppm HCl. Formation of 4-chlorophenol reached a maximum at 600°C, a value in accord with Eklund et al. (8). At this temperature, the effect of HCl concentration on phenol conversion was determined. Figure 5 shows the relationship between phenol conversion and HCl concentration, on a double logarithmic scale. Control testing with phenol but without HCl produced negative results for chlorinated species formation.

Tests with HCl also produced measurable amounts of non-chlorinated dibenzo-p-dioxin (DBD) and dibenzofuran (DBF). Figure 6 shows DBD and DBF formation (GC trace peak area), normalized by the concentration of phenol fed, as a function of temperature at 3000 ppm HCl. The maximum production occurs at temperatures similar to the 4-chlorophenol peak, around 600°C. Between the HCl concentrations of 200 and 4600 ppm, no effect of HCl concentration on DBD or DBF formation levels is noticeable (Figure 7). Control testing with phenol but without HCl did not yield any detectable DBD/DBF.

The previous tests were repeated using Cl_2 gas instead of HCl. Numerous mono- and multi-substituted chlorinated aromatics were found, as indicated by a representative GC trace (Figure 8). These species include 4-chlorophenol; 2,4-dichlorophenol; 2,6-dichlorophenol; 2,4,6-trichlorophenol; pentachlorobenzene; and hexachlorobenzene. Unspecified peaks in this and subsequent GC traces are generally non-chlorinated, non-aromatic hydrocarbons. The substitution pattern of Cl is reasonable: the OH^- group of phenol, $C_6H_{11}OH$, is a strong activator for electrophilic aromatic substitution reactions with halogens, giving products with halogens in the ortho (2,6) and/or para (4) positions. The change in chlorinated aromatic product concentrations as a function of temperature (200 to 800°C) is shown in Figure 9, indicating a maximum conversion of phenol around 350°C. This temperature of maximum phenol conversion compares with the 600°C value for HCl. More importantly, the amounts of phenol conversion are at least five orders of magnitude higher with Cl_2 than with HCl. Product formation at 400°C, near the temperature of maximum formation, is proportional to Cl_2 concentration (Figure 10). Control tests with phenol but without Cl_2 did not form any chlorinated organics. Tests with Cl_2 did not yield measurable amounts of DBD or DBF, as with HCl.

The formation of chlorinated species was examined during room temperature experiments to investigate the potential for post-reactor chlorination. Significant amounts of phenol conversion resulted with Cl_2 (none with HCl), lending difficulties to sampling procedures and quantitative determinations. Ambient temperature tests resulted in approximately 25% of the total chloroaromatics formation noted in high temperature experiments.

Summary

HCl gas appears to play some role in formation of PCDD/PCDF precursors, such as DBD and DBF, but its involvement as the principal source of chlorine for chlorinated aromatics appears unlikely. The formation of some chlorinated organics (Figure 3) during gas-phase reaction of HCl and phenol validates homogeneous phase findings of Eklund et al. (7). While experiments with Cl_2 gas resulted in significant quantities of mono- and multi-substituted chlorophenols, no trace of PCDD or PCDF formation was ever observed. The absence of gas-phase formation of PCDDs and PCDFs agrees with theoretical determinations of Shaub and Tsang (10) using equilibrium values and estimates of the kinetic reaction parameters. Nonetheless, while ultimate formation of PCDD/PCDF must depend on some other step, a definite effect of Cl_2 gas (and possibly limited involvement of HCl) in forming chlorinated precursors has been demonstrated.

Heterogeneous Phase Formation

Given that Cl_2 reacts to a much greater extent with phenol than HCl, as seen in the homogeneous tests, the potential involvement of a solid-catalyzed reaction producing Cl_2, such as the Deacon process, was investigated. Tilly (25) summarizes the reactions in the Deacon process involving copper chloride (CuCl), a compound often present in incinerator fly ash:

$$2\ CuCl + 1/2\ O_2 \longleftrightarrow Cu_2OCl_2$$

$$Cu_2OCl_2 + 2\ HCl \longleftrightarrow 2\ CuCl + H_2O + Cl_2.$$

To examine this mechanism, a quartz wool plug was inserted into the COR with approximately 1 g of a dispersed, solid catalyst (CuCl). Tests were run around 450°C, a temperature reported by Tilly to have yielded approximately 40% conversion of HCl to Cl_2. Analyses showed formation of multichlorinated phenols and PCDDs (Figure 11) after 1000 ppm HCl and phenol were passed through the bed for 1 h between 350 and 500°C. This supports the Deacon process mechanism in which the product, Cl_2, chlorinates the phenol. The products included 4-chlorophenol; 2,4-dichlorophenol; 2,6-dichlorophenol; 2,4,6-trichlorophenol, and several PCDDs (di-, tri-, and tetrachlorinated).

Control tests were conducted, as above, with CuCl and gaseous phenol but without HCl. GC analysis (Figure 12) showed production of similar multi-chlorinated phenols and PCDDs, identical to the results from tests with HCl, over the temperature range of 350 to 500°C. This was contrary to expectations, suggesting that the chlorine source is derived from the CuCl and that the gas-phase HCl concentration is irrelevant to phenol chlorination.

This also raises doubts about the suggestion that HCl provides the initial chlorine source for organic chlorination via the Deacon process. Further tests were done to identify the source of chlorine for phenol chlorination.

Tests with CuCl and phenol but without HCl were repeated and analyzed for Cl_2 by the KI test. Evolution of Cl_2 gas from the CuCl within the quartz wool bed was monitored by a time-resolved KI test. Results, presented in Figure 13, show that Cl_2 gas was emitted from CuCl experiments at 450°C. Production of Cl_2 was highest initially, then trailed off to a relatively constant level, around 50 ppm. Subsequent introduction of 1000 ppm HCl gas flow resulted in a step increase in Cl_2 production to 200 ppm (600 ppm HCl emission), remaining constant thereafter. XRD analysis of the quartz wool plug showed the presence of Cu_2OCl_2, an intermediate product in the Deacon process. These two analyses verified that the Deacon process was indeed occurring with the HCl reactant.

Testing with the emission levels as above (200 ppm Cl_2 and 600 ppm HCl), but without CuCl in the quartz wool, gave a GC trace analysis (Figure 14) that yielded most of the same types of chlorophenols and DBD/DBF seen in the CuCl tests, albeit at smaller quantities and without PCDDs. This suggests that the presence of the catalyst does alter the product yield. It is possible that the CuCl, or a derivative thereof, acts catalytically to make more precursors beyond those which can be expected solely from non-catalytic, homogeneous tests. It is also likely that CuCl may participate in a reaction between precursors and the dioxins themselves.

The above tests were repeated for a longer duration, 15 h, after which Cl_2 emission levels dropped to only 5 ppm. XRD product analysis of the catalyst plug showed Cu, Cu_2O, and trace quantities of CuCl. Subsequent HCl addition (1000 ppm) yielded only 11 ppm of Cl_2, indicating that the virtual absence of CuCl in the decomposed catalyst prevented the Deacon process from recurring. Predictably, after the 15 h course was run, addition of phenol without HCl resulted in product formation (see Figure 15) substantially less than for the undecomposed catalyst tests.

The evolution of Cl_2 gas from 1) CuCl decomposition in the absence of HCl and 2) initiation of the Deacon process by flowing HCl over the partially decomposed CuCl catalyst, complicates the analysis, raising the question of the actual chlorine source in municipal waste incineration for precursor chlorination. Four potential paths were examined.

The possibility of reaction complicated by the involvement of SiO_2, present in the quartz reactor tubes and quartz wool, has been proposed by Clews and Thompson (22) and Shaub and Tsang (11). They suggested formation of Cl_2 by:

$$CuCl + SiO_2 + 1/2\ O_2 \longleftrightarrow 1/2\ Cl_2 + CuOSiO_2$$

with subsequent reaction of the Cl_2 to form chlorinated species. Our work tested this hypothesis by examining the CuCl-laden quartz wool plug after testing with HCl and phenol (conditions in which PCDD formation was noted).

The quartz wool material was analyzed by XRD for the presence of copper silicate ($CuOSiO_2$) reaction products and found to be negative for $CuOSiO_2$, lending doubt to this hypothesis.

A second possibility for Cl_2 formation (25) follows the mechanism:

$$2\,CuCl + 1/2\,O_2 \longleftrightarrow Cu_2OCl_2$$

$$Cu_2OCl_2 + 1/2\,O_2 \longleftrightarrow 2\,CuO + Cl_2.$$

The product of the first reaction was found by XRD analysis of the CuCl-laden plug under both HCl and non-HCl conditions. However, when CuCl (with only the basic process gases) was run for 15 h and the catalyst-laden plug analyzed by XRD, the products identified included elemental copper (Cu), Cu_2O, and trace CuCl. However, the presence of Cu_2O as opposed to CuO, suggested a more direct breakdown of Cu_2OCl_2 (the third possibility):

$$Cu_2OCl_2 \longleftrightarrow Cu_2O + Cl_2.$$

The detection of all of these species by XRD or KI tests maintains the possibility of this mechanism, although its validity remains untested.

The presence of Cu in the reactor plug might indicate a fourth chlorination scheme, not involving Cl_2. The fourth possibility is a direct reaction:

$$CuCl + \text{phenol} \longleftrightarrow \text{chlorophenols}$$

which does not require a mechanism for Cl_2 formation. This possibility was examined by CuCl tests without the presence of O_2 at 450°C. KI tests proved no emission of Cl_2 since lack of oxygen prevented the Deacon intermediate. Phenol was introduced into the reactor, and GC/FID analysis (Figure 16) found no chlorinated species, suggesting that direct reaction does not occur.

This further suggests that, for the heterogeneous tests, chlorination of the phenol occurred through reaction with Cl_2, from either breakdown of CuCl or the Deacon process. This suggestion accounts for results of Vogg et al. (18) who found increased PCDD/PCDF levels in fly ash heated in flowing air for 2 h at 300°C. Decomposition of metal chlorides present in the fly ash may have produced Cl_2, resulting in subsequent chlorination of residual organic precursors to form PCDD/PCDF. When they added HCl or flue gas, they found even greater levels of PCDD/PCDF, suggesting the occurrence of the Deacon process with the metal chloride catalysts.

This suggests that, in an incinerator environment, precursors to PCDD/PCDF may be formed through either gas-phase homogeneous reactions with organics and Cl_2 or a heterogeneous, catalytic reaction between organics and Cl_2.

Summary

These results imply a formation mechanism in incinerators involving reaction between organics and Cl_2 to form PCDD/PCDF precursors. The Cl_2 is formed either through conversion of HCl by the Deacon process or through side reactions involving CuCl. Controlling the levels of Cl_2 and/or HCl (prior to conversion to Cl_2) in a temperature zone above that of PCDD/PCDF formation may significantly reduce PCDD/PCDF levels in the incinerator emissions. The large body of literature on high temperature control of sulfur dioxide (SO_2) by injection of calcium-based (Ca) sorbents into coal-fired combustors suggests that this technology might be applied similarly to HCl/Cl_2 control. The sorbent would be injected into the flue gas where it would react to remove both HCl and Cl_2 as a stable solid product before PCDD/PCDF formation could occur.

SORBENT INJECTION STUDY

RESULTS AND DISCUSSION

HCl and Cl_2 Removal

Tests were run in the STDR to determine the ability of commercial $Ca(OH)_2$ sorbents to remove gaseous HCl and Cl_2 from simulated flue gas streams. Time-resolved experiments, ranging from 0.3 to 5 s, were run at sample temperatures of 400 and 800°C. Figures 17 and 18 show the rate of sorbent reaction with HCl and Cl_2, respectively, at both temperatures. The rate of sorbent conversion is similar at both temperatures for HCl. For Cl_2, sorbent conversion is greater at 800°C.

The reaction product (from an 800°C HCl test) was analyzed by XRD and found to be calcium chloride hydroxide [$CaCl_2 \cdot Ca(OH)_2$], JCPDS registry No. 36-983. Rapid hydration of the sample from the $CaCl_2$ form is likely to have resulted in the major product identified by XRD. The important point of this result is the Ca/Cl molar ratio, 1/2, in the product. This indicates the amount of chlorine capture expected from a given amount of sorbent conversion.

Summary

A representative rate of reaction for HCl (20% in 1 s) depletes 5% of the STDR HCl source, a value close to that of differential operation. Thus, if sorbent were injected at 800°C with a Ca/Cl ratio of 2/1, HCl removals approaching 80% and Cl_2 removals approaching 40% could reasonably be expected in 1 s. This preliminary testing shows that reaction occurs over a wide range of temperature (400 to 800°C). This suggests that over a wider temperature range, an effective sorbent residence time of even greater than 1 s is likely, leading to even higher Cl capture levels.

CONCLUSIONS

PCDDs and PCDFs have been formed from low temperature (350°C) reactions between Cl_2 and phenol precursors. Both homogeneous gas-phase and heterogeneous catalytic reactions have been observed to produce chlorinated species, including PCDDs and PCDFs. While the presence of HCl is not likely to be directly responsible for PCDD/PCDF formation, at elevated temperatures (650°C) it may promote the formation of DBD and DBF (PCDD/PCDF precursors) and participate in the Deacon process, forming Cl_2 for the subsequent chlorination reactions. Removal of the HCl prior to the 350°C region by injection of calcium-based sorbents and formation of $CaCl_2$ solids could remove the chlorination source and likely prevent formation of some aromatic precursors. Reduction in formation levels of toxic PCDDs and PCDFs are likely, therefore, through removal of the chlorination source prior to optimum PCDD/PCDF reaction temperatures accompanied by proper control of combustion conditions to minimize formation of organic precursors.

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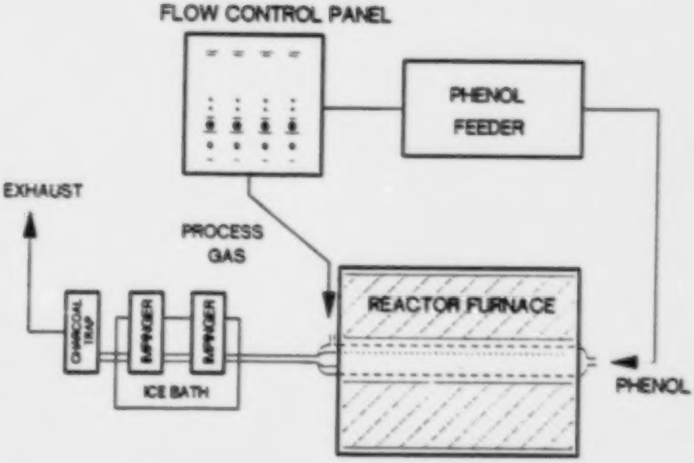
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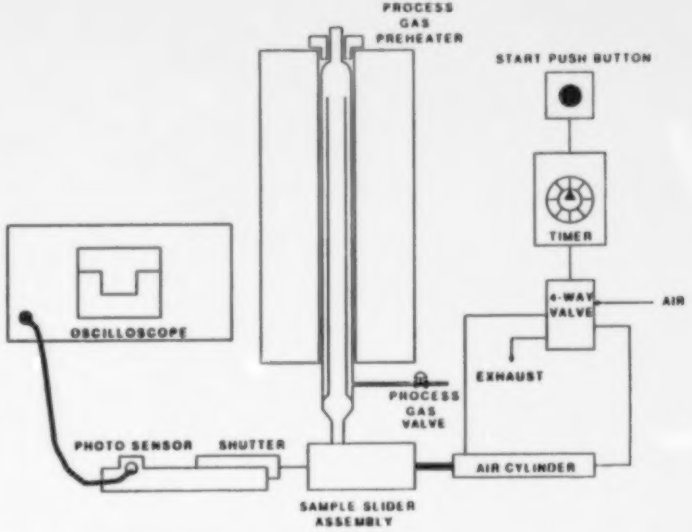
PHENOL
FEEDER
EXHAUST
PROCESS
GAS
CHARCOAL TRAP
IMPINGER
IMPINGER
ICE BATH
REACTOR FURNACE
PHENOL

Figure 1. The COR experimental apparatus.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GAS
PREHEATER
START PUSH BUTTON
TIMER
4-WAY
VALVE
AIR
OSCILLOSCOPE
EXHAUST
PROCESS
GAS
VALVE
PHOTO SENSOR
SHUTTER
AIR CYLINDER
SAMPLE SLIDER
ASSEMBLY

Figure 2. The STDR experimental apparatus.

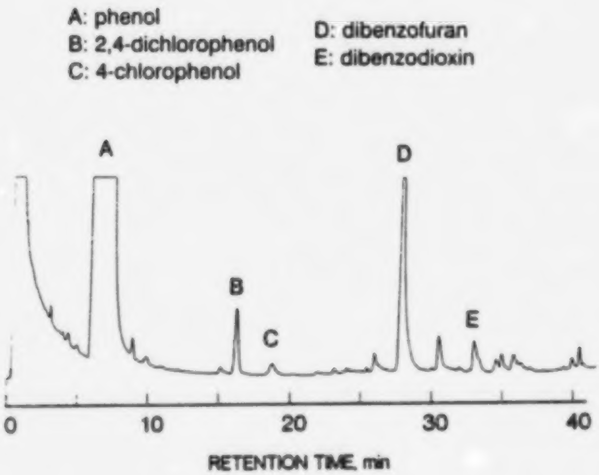

Figure 3. The GC trace for 1000 ppm HCl, 1000 ppm phenol, and basic process gases at 600°C, identifying principal peaks.

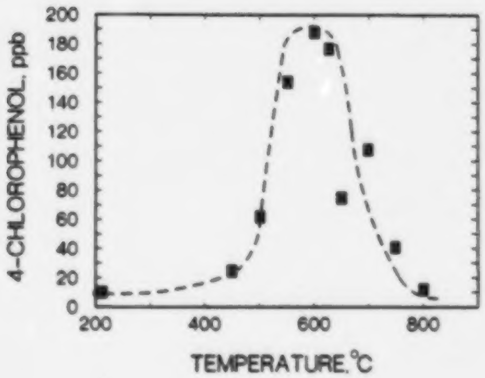

Figure 4. Temperature dependency of 4-chlorophenol formation at 3000 ppm HCl.

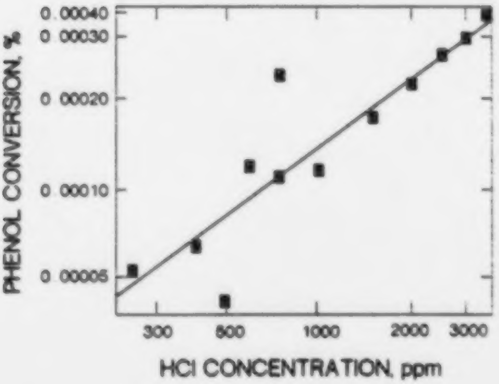

Figure 5. HCl concentration dependency of phenol conversion at 600°C.

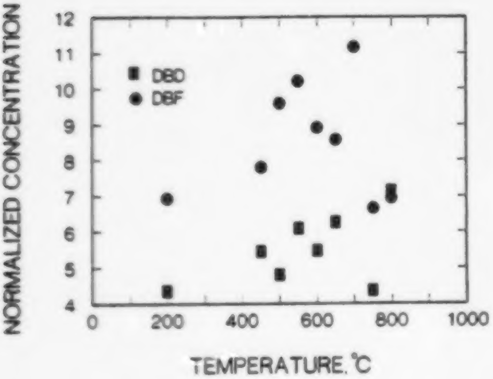

Figure 6. Temperature dependency of DBD and DBF formation with 3000 ppm HCl.

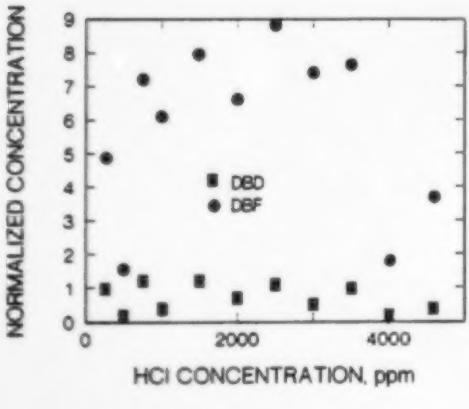

Figure 7. HCl concentration dependency of DBD and DBF formation at 600°C.

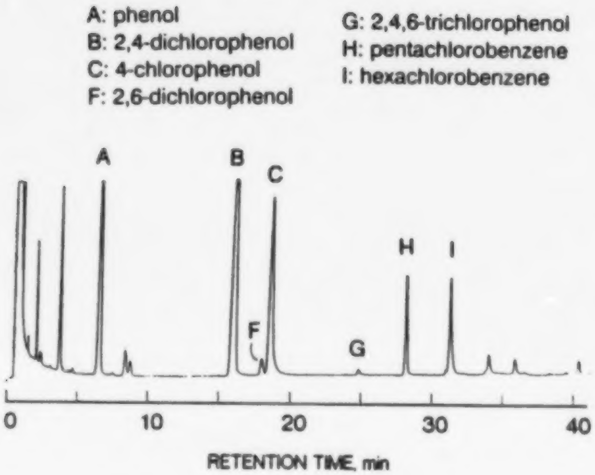

Figure 8. The GC trace for 1000 ppm Cl_2, 1000 ppm phenol, and basic process gases at 400°C, identifying principal peaks.

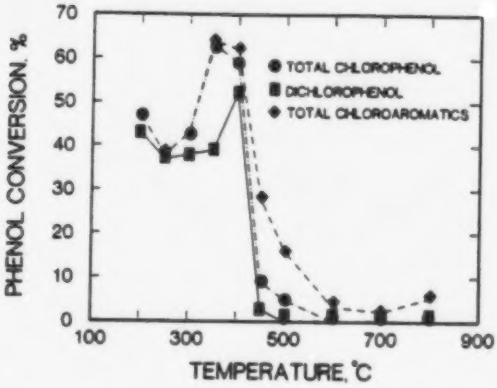

Figure 9. Temperature dependency of chlorinated aromatics formation with 1000 ppm Cl_2.

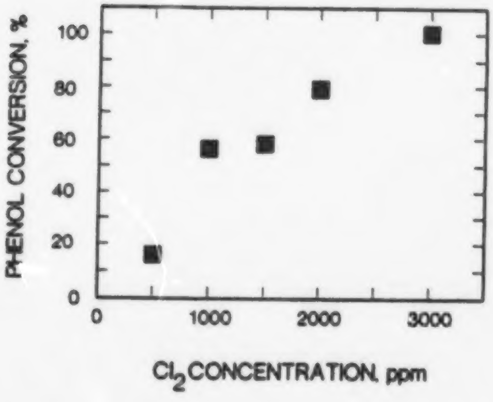

Figure 10. Cl_2 concentration dependency of phenol conversion at 400°C.

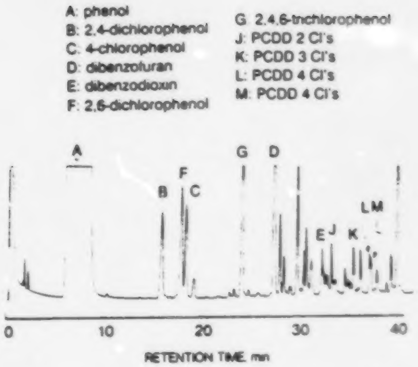

Figure 11. The GC trace for 1000 ppm HCl, 1000 ppm phenol, and basic process gases at 450°C with a CuCl catalyst bed, identifying principal peaks.

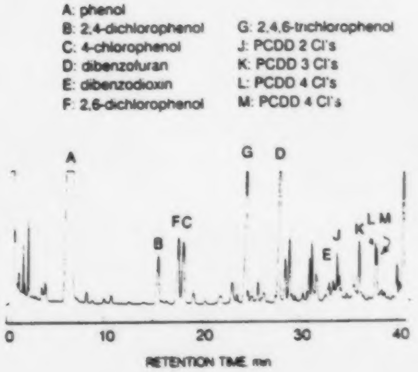

Figure 12. The GC trace for conditions of Figure 11 without HCl.

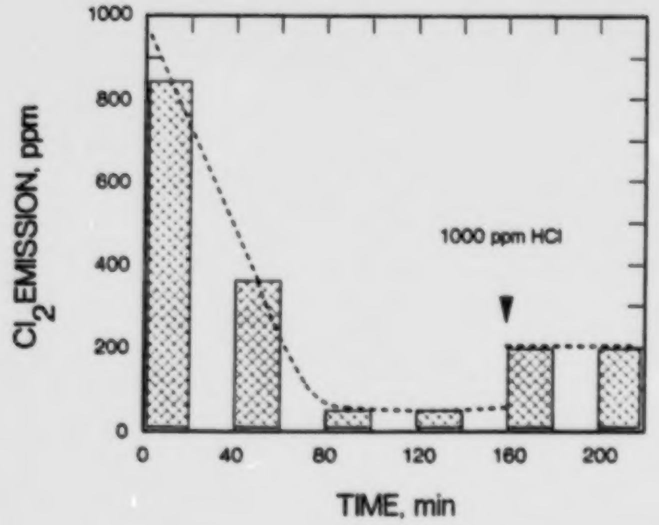

Figure 13. KI test yield of Cl_2 emissions over time from CuCl and basic process gas at 450°C.

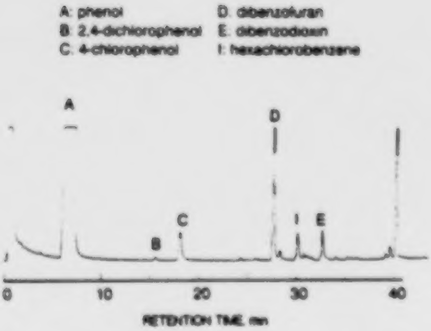

Figure 14. The GC trace for 600 ppm HCl, 200 ppm Cl_2, 1000 ppm phenol, and basic process gases with a quartz wool plug.

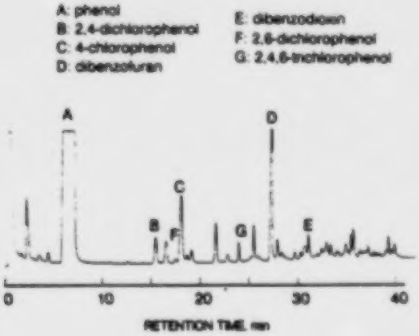

Figure 15. The GC trace for CuCl catalyst with 1000 ppm phenol and basic process gases at 450°C in the absence of HCl.

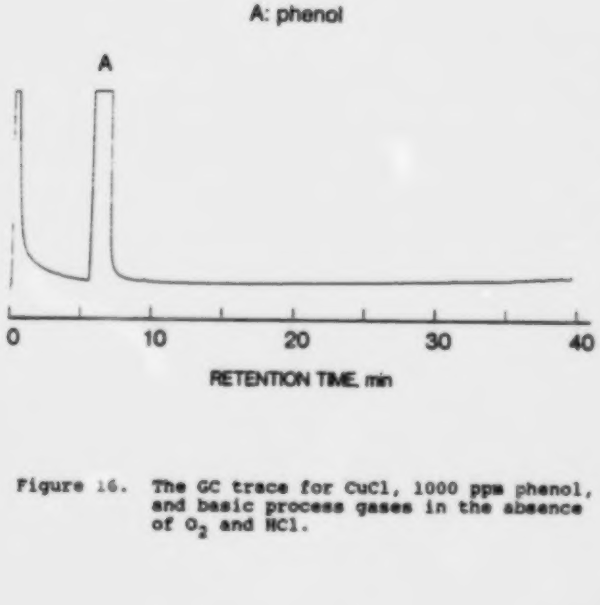

Figure 16. The GC trace for CuCl, 1000 ppm phenol, and basic process gases in the absence of O_2 and HCl.

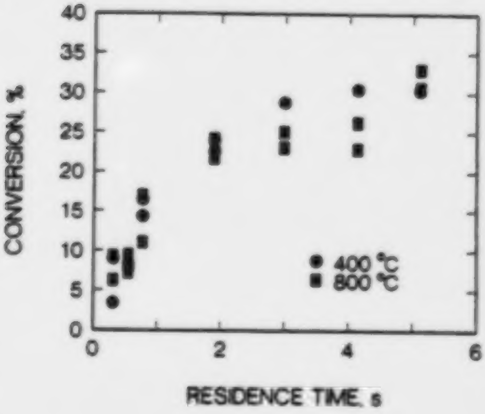

Figure 17. Sorbent conversion of $Ca(OH)_2$ and HCl versus time at 400 and 800°C.

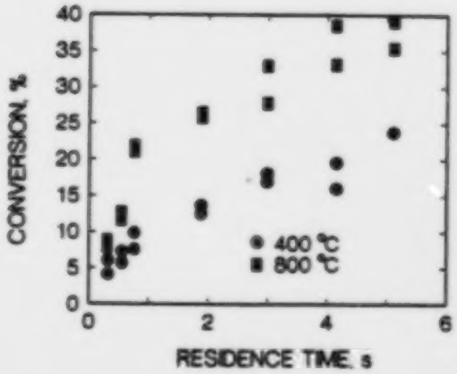

Figure 18. Sorbent conversion of $Ca(OH)_2$ and Cl_2 versus time at 400 and 800°C.

REDUCING ACID GAS EMISSIONS FROM MASS-BURN WASTE-TO-ENERGY PLANTS BY DIRECT LIME INJECTION

by:

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ABSTRACT

With the introduction of new environmental protection legislation, federal and local air emission regulations continue to become more stringent, especially in the area of reducing acid gas emissions from power generating facilities. Currently, the most widely used methods for control of acid gas emissions are low temperature wet or dry gas scrubbers.

Direct injection of alkali sorbents into the hot coal- or waste-fired furnaces has been the subject of intense investigation in the past.

The direct injection of lime sorbents into hot waste-fired furnaces was studied in a mass-burn waste-to-energy plant which uses the Westinghouse O'Connor combustor technology to burn municipal solid waste. The test results reveal SO_2 and HCl removal efficiencies that are promising. The emissions of other gaseous and trace metal pollutants were also measured during the test, and their concentration levels did not change appreciably with sorbent addition. The fly ash carryover rate increased with hydrated lime addition.

INTRODUCTION

In conventional coal- or waste-fired power generating plants, low temperature post combustion wet or dry flue gas desulfurization processes (FGDs) are usually employed in order to comply with clean air laws and New Source Performance Standards. Pre-combustion and in-situ techniques for reducing the emission of acidic gases are also being developed by government and utility funded research organizations.

Wet scrubbing, by water spraying alone, is usually sufficient for control of HCl and HF emissions. However, for effective removal of SO_2 an alkali sorbent must be used. In dry scrubbing, powdered alkaline sorbents are injected in ducts or especially designed vessels to capture the acidic gases by direct contact.

Low temperature wet scrubbing technologies suffer from a number of operational difficulties. Buildup of insoluble salts on the walls of the scrubber vessel (scaling), corrosion and fouling of the vessel internals, difficulties in sedimentation of the produced salts, and clarification and recycling of scrubber liquor are among the most common problems encountered in wet scrubbing systems.

In low temperature dry scrubbing systems much of the above scaling, sedimentation and wet solid waste handling problems are eliminated at the expense of higher operational and maintenance costs. The acid removal efficiency of the dry scrubbing systems depends on the degree of intimate mixing between the dry sorbent powder, usually hydrated lime ($Ca(OH)_2$), and the flue gas. Optimum design of the sorbent injection system and the gas-solid contacting vessel, therefore, become very important.

In semi-dry scrubbing systems the spraying dynamics of the sorbent is enhanced by injecting the sorbent as a slurry. The sorbent particles dry quickly as they come in contact with the hot gases and form dry reaction products (calcium salts). More elaborate hybrid combinations of wet and dry scrubbing technologies have also been developed and are currently used in large and small power plants.

The low temperature scrubbing systems require large vessels and extensive lime storage, handling and slaking equipments. The size and, therefore, the cost of the these systems are a function of the type of the fuel being fired, the amount of flue gas to be processed, the level of acidic gases present in the flue gas stream, and the level of acid gas removal desired. Less expensive pre-combustion and in-situ cleanup processes that can accomplish some degree of sulfur removal or acid gas control can, therefore, reduce the operational complexity and the capital cost of the low temperature scrubbing systems substantially.

Pre-combustion physical and chemical coal cleaning methods, such as by coal washing and caustic treatment methods, (1, 2), cannot be simply used for removing sulfur from raw municipal solid waste. Processes for direct, in-situ, removal of acidic gases by high temperature injection of sorbents into furnaces and boilers are being actively developed for use in both waste- or fossil fuel-fired power plants.

Two of the most promising developments are the Lime Injection Multistage Burner technology (the LIMB (3,4) process) and the fluidized bed combustor systems (5). When burning high-sulfur fuel, an acid gas polishing scrubber is often used at the tail end of the process to ensure compliance to air emission standards at all times. Fabric filters or efficient Electrostatic Precipitators (ESP's) remove the particulates leaving the post combustion boiler sections.

The acid gas removal efficiency of the direct high temperature sorbent injection processes is a strong function of the chemical and physical properties of the sorbent particles, the degree of intimate mixing between the sorbent particles and the flue gas, the time and temperature history of the sorbent particles and the flue gas, and the sorbent feed rate.

In the following sections the results of a series of exploratory tests conducted by Westinghouse Electric Corp. to evaluate the effectiveness of high temperature acid gas removal in mass-burn waste-to-energy plants by the direct lime injection will be presented. The tests were conducted at the Bay County, Florida mass-burn waste-to-energy plant.

BAY COUNTY FACILITY PROCESS DESCRIPTION

The main components of the Bay County waste-to-energy plant are schematically shown in Figure 1. Two Westinghouse/O'Connor combustors, each coupled to a heat recovery boiler, are used for burning MSW to generate steam and, ultimately, electricity.

The O'Connor combustor is a rotary steel drum combustor made of steel water tubes joined to one another by welded steel membranes called webs. Cooling water is circulated through the water tubes to cool the combustor walls and maintain long term integrity. MSW is first pushed onto an inclined travelling conveyor by a front-end loader at the tipping floor level. The feed material is then conveyed into the feed hopper. Hydraulic rams are used to feed the material into the combustor. Preheated combustion air is forced into the combustor through the perforations in the combustor webs. The transport of the waste through the combustor is achieved by controlling the rotational speed of the combustor.

The rotary combustor is connected to a water-wall boiler which extracts heat from the combustion flue gases to generate steam. Heat in the combustion flue gases is absorbed by the water-wall boiler, the superheater, the convection section, and the air heater before the flue gas enters the air pollution control equipment and stack. At the Bay County plant, the fly ash that is carried over by the flue gases is collected from the superheater, and convection section hoppers, and the electrostatic precipitator hoppers. Opacity, CO, and O_2 emission levels are monitored continuously by the plant instruments for combustion control and emission monitoring purposes. The generated steam is used to drive a turbine generator and produce electricity. The combustor bottom ash and the collected fly ash are quenched in an quench tank and later deposited into dumpster trucks by the ash drag conveyor. The combined ash is disposed in a lined landfill operated by the County.

EXPERIMENTAL RESULTS

Due to the exploratory nature of the tests, the sorbents were manually fed to the combustor at the MSW feed hopper opening. This crude method of lime addition may have in fact led to lower acid gas removal due to the initial exposure of the lime particles to the flaming regions inside the combustor.

A number of plant instruments were used to continuously record the the plant operating data during the tests. Continuous emission monitoring equipment and manual U.S. EPA approved sampling and analysis methods were also used to measure gaseous effluent pollutant concentration levels.

The test series was divided into a number of runs, hereafter identified as set points 2A to 2D, 8A and 8AR, 8B and 8BR, and 8C. Four days of testing were conducted burning only municipal solid waste (MSW) (set points 2A to 2D). This was done to obtain baseline emission information. Four days of testing were conducted by adding controlled amounts of hydrated lime (set points 8A and 8AR), quicklime (set points 8B and 8BR) and crushed limestone (set point 8C) into the rotary O'Connor combustor together with the incoming MSW.

The crushed limestone test was conducted to see whether lime calcination and acid gas removal can be performed in a single process. Quicklime is the product of calcination of limestone and when reacted with water it disintegrates, i.e. pops, into small hydrated lime particles. Hydrated lime is more expensive than Quicklime, but, it is generally found to be more effective for acid gas removal due to its very high specific surface area. Limstone is the least expensive material, but it must be calcined to become reactive. The mean particle size of the quicklime and the crushed limestone particles used during the test were 640 and 1600 microns respectively. The hydrated lime was delivered in pulverized form, with 99.5 percent of the particles being less than 45 microns in diameter.

Table 1 summarizes the actual test operating conditions and some of the process data collected during the test.

Most of the flue gas pollutant levels were measured at sampling ports in the duct work before the plant's ESP in order to eliminate the effect of any possible cleanup or condensation in the ESP. The calculated particulate removal efficiency of the ESP for both the lime addition and the MSW-only set points are presented in Table 2.

Flue gas CO, SO_2, NOx, $SO_3^=$, HCl and HF concentrations are presented in Tables 3 and 4. The CO, SO_2 and NOx concentrations (Table 3) were measured by continuous emission monitoring instruments, while the $SO_3^=$, HCl, and the HF levels were measured manually by using U.S. EPA approved reference methods and equipment. The $SO_3^=$ and HF levels (Table 3) were measured at the stack due to unavailability of port locations at the ESP inlet.

The concentration of trace metals on the fly ash particulates were measured at the inlet to the ESP. The data for six metals is presented in Table 5.

DISCUSSION OF RESULTS

It is widely known that metal oxides can capture sulfur and chlorine species over specific temperature ranges. Calcium based sorbents are widely used in low temperature acid gas scrubbers and are also known to react with acidic gases at high temperatures. For example, in the LIMB process lime is injected to react with SO_2 at temperatures as high as 2400 °F, (5). In fact, at temperatures higher than approximately 930°F and in presence of oxygen, calcium oxide can react with sulfur dioxide to produce calcium sulfate (6), an environmentally inert and stable material.

The effectiveness of the direct dry lime injection processes at high temperatures is influenced by many factors. The properties of the raw sorbent that lead to better pore structure development during the calcination process and the size of the sorbent particles; the sorbent feed rate and the acid gas concentration levels; the type of fuel fired, the furnace operating conditions; the time and temperature history of the flue gas and sorbent particles; the degree of intimate mixing between the sorbent and the furnace gases can all impact the acid gas removal efficiency of these processes.

As mentioned earlier, the sorbent material was injected into the combustor at the MSW feed hopper opening. Therefore, the sorbents entered the combustor at the beginning of the zone one, as depicted in Figure 2. On Figure 2 the estimated time and temperature profile of the flue gas inside the hot sections of the incinerator units installed at the Bay County plant are displayed.

The summary process data presented in Table 1 indicate that during the MSW test the higher heating value (HHV) of the MSW fuel burned by the plant had varied from 3760 BTU/lb to around 5100 BTU/lb. This type of fuel variability is commonly observed in waste-to-energy plants. The HHV of the MSW portion of the feed stream was not measured by actual sampling of the raw MSW. This was due to the general difficulty in obtaining representative MSW samples and the difficulty in preparation and analysis of bulky MSW samples by standard calorimetric methods. To overcome this problem the hybrid boiler-as-a-calorimeter test procedure was used to calculate the average HHV of the MSW. This is accomplished by accounting for the process heat outputs, heat losses and credits, and measuring the fuel feed rate. In the calculations, the moisture content of the MSW was also assumed to be around 28% based on past experience.

The data presented in Table 1 also indicate that the combustion efficiency did not deteriorate as a result of sorbent addition. The only noticeable, and expected, change is an increase in the amount of fly ash carried out of the boiler by the flue gas during the hydrated lime set point.

The ESP efficiency data (Table 2) show that in spite of the observed increased fly ash carryover rate into the ESP, the collection efficiency of the ESP did not drop. This is probably because of the high moisture content of the flue gas from MSW incinerators (around 14 to 18%) which helps reduce the affect of the increased fly ash resistivity due to presence of the dry sorbent. In coal-fired plants moisture is added to the flue gas stream upstream of the ESP to reduce the increased resistivity of the lime rich fly ash.

Process and emission data from the limestone set point (set point 8C) is not presented. This is because during the initial few hours of this test no noticeable reduction in the flue gas SO_2 levels were noticed. The test was terminated and no detailed data was saved.

The overall set point flue gas SO_2, $SO_3^=$ and HF levels (Table 3) show a definite improvement with quicklime and hydrated lime addition. The average set point NOx emissions are seen to remain essentially the same, i.e. below 130 ppm, with or without lime addition. The reduced CO emission levels is probably due to the better operation of the incinerator by the plant operators rather than any kinetic effects by the sorbents.

During set points 8A and 8B it became suspect that sorbent fines may have migrated into the sampling lines of the continuous emission monitoring system and caused the observed low flue gas SO_2 levels. Set points 8AR and 8BR were, therefore, run to confirm the set point 8A and 8B results. In preparation for these repeat set points, flue gas SO_2 samples were taken both manually and continuously and compared to each other. Gas samples were extracted both from the same sampling probe and via separate probes placed close to each other. This redundant sampling scheme was repeated during the actual test periods. The flue gas SO_2 concentrations measured during set points 8AR and 8BR are presented in Figures 3, 4 and 5.

The plot of the SO_2 levels measured during set point 8BR indicate that there was a 40 minute time lag from the time hydrated lime was added at the feed hopper opening to the time the SO_2 levels at the inlet to the ESP started dropping appreciably. Similar observations were made during the set point preparation activities. Consequently, in order to ensure that the effects of any initial sorbent buildup, if any, are eliminated from the measurements, the manual sampling period was adjusted to between 20 to 30 minutes . The manually measured SO_2 levels are illustrated with solid horizontal lines in Figures 3, 4 and 5. The length of the solid lines indicates the duration of the sample extraction period. The close agreement between the manually and continuously measured SO_2 levels is evident.

The plot of the flue gas SO_2 levels measured during the quicklime set point (Figure 5) suggests that although an overall average reduction in SO_2 levels is achieved, the actual emission levels can vary over a broad range.

The actual average flue gas SO_2 and HCl concentration values measured during set points 8AR and 8BR are presented in Table 4. In order to calculate the set point SO_2 and HCl removal efficiencies, the emission levels measured during set points 2A to 2D were averaged to establish baseline data, (144.5 and 509 ppmdv at 7% O_2). The calcium to sulfur and chlorine mole ratios (Table 4) are calculated on the basis of these baseline concentrations and the amount of sorbent actually fed to the combustor.

The removal efficiency is found out to be higher with hydrated lime addition (around 60 to 88% for SO_2 and 25 to 76% for HCl) than with quicklime addition (20 to 73% for SO_2 and 34 to 66% for HCl). In high temperature acid gas scrubbing applications, hydrated lime is generally found to be more reactive than quicklime. This is because when calcium oxide is produced during dehydration of hydrated lime it becomes more porous than pure calcium oxide. In addition, the smaller hydrated lime particles (less than 45 micron size) have a much larger specific surface area than the larger quicklime particles (640 micron). No appreciable SO_2 or HCl removal was detected during the crushed limestone set point.

The data presented in Table 4 demonstrates that a wide range of acid gas removal efficiencies should be expected with direct lime addition processes. This is to be expected in mass-burn waste-to-energy plants where temperature, sulfur and chlorine concentration excursions are common, and it is nearly impossible to accurately control the calcium to sulfur mole ratio. In commercial applications it will, therefore, be necessary to install a low temperature flue gas polishing system to ensure compliance with air emission requirements at all times. In such applications it would be possible to pass only part of the treated flue gas through the low temperature polishing system.

The addition of hydrated lime or quicklime did not appreciably change the level of trace metals on the carryover fly ash particles (Table 5).

SUMMARY AND CONCLUSION

Generally, the test results indicate that promising acid gas removal rates can be achieved by direct injection of hydrated lime and quicklime into mass-burn MSW-fired furnaces. The tests were, however, conducted in a waste-to-energy plant which uses the rotary water-cooled O'Connor combustor technology. The lime sorbent was injected at the front end of the combustor, and had, therefore, been exposed to very hot temperature zones. The possibility of some sorbent deactivation, due to partial dead-burning and sintering of some of the sorbent particles, cannot be ignored.

Much detailed laboratory and field testing still remain to be done, especially in the areas of sorbent injection location, sorbent selection, sorbent feed rate and performance optimization, before the process can be viably used commercially.

Results from sorbent characterization and screening tests, conducted in-house at Westinghouse, have confirmed the findings of the current field tests. For example, it is learned that limestone and dolomite must be calcined before injection into the incinerator to be effective. Both calcium oxide and calcium hydroxide have been found to be effective for SO_2 and HCl removal in the 800^{o}F to 2000^{o}F temperature range.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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TABLE 1. SUMMARY OF AS-RUN OPERATING CONDITIONS

Parameters	SET POINT							
	2A	2B	2C	2D	8AR	8A	8B	8BR
Feed Streams								
o MSW Flow, TPD	370	261	NM	268	NM	298	340	NM
- HHV[a] , BTU/lb	3760	5104	---	4463	----	4241	3881	---
o Sorbent Type					Hydrated Lime		Quick Lime	
o Sorbent Feed, TPD	0	0	0	0	3	3	3.4	2.64
Steam								
- Flow Rate, klb/hr	69.9	71.8	63.5	63.5	71	70.6	69.5	70
- Temperature, °F	772	756	761	751	751	749	756	749
- Pressure, psig	568	570	565	565	565	566	565	566
Ash								
o Bottom Ash								
- Flow Rate ,TPD(dry)	97.5	85.1	NM	49.6	NM	64.6	69.6	NM
- HHV[b] , BTU/lb	540	430	1,032	394	NM	232	376	NM
o Fly Ash								
- Flow Rate[c],lb/hr.	NM	711	NM	602	NM	1,002	726	NM
Combustion[d] Efficiency % (X100)	99.92	99.78	99.87	99.72	99.95	99.92	99.88	99.94

NM: Not Measured.

[a] Calculated value by the Boiler-as-a-Calorimeter Method.

[b] Dry basis.

[c] Fly ash captured in the ESP and airheater hoppers.

[d] C.E. = $(CO_2)/(CO_2+CO)$. CO @ 12% CO_2.

TABLE 2. ESP PARTICULATE REMOVAL EFFICIENCY

Set Point	ESP Efficiency (%)
2A	98.7
2B	98.0
8B	98.5
8A	99.0

TABLE 3. AVERAGE FLUE GAS CO, SOx, NOx and HF CONCENTRATIONS MEASURED AT THE ESP INLET (ppmdv @ 7% O_2)

Constituent Concentration	Set Point								
	2A	2B	2C	2D	AVG(2A-2D)	8B	8A	8AR	8BR
CO									
Minimum	70	76	97	194	109	60	58	52	64
Average	103	270	196	313	220	157	102	64	76
Maximum	142	924	384	514	491	285	173	81	93
SO_2									
Minimum	86	114	83	86	92	37	23	16	51
Average	127	183	158	110	144	73	35	40	84
Maximum	205	324	315	185	257	196	39	65	116
NOx									
Minimum	107	94	95	110	101	115	106	94	107
Average	114	113	121	120	117	130	118	101	120
Maximum	124	140	156	133	138	154	125	104	132
SO_3* [a]	NM	NM	3.61	3.63	3.62	2.87	1.85	NM	NM
HF (gas)[a]	NM	NM	1.43	1.38	1.40	1.09	0.77	NM	NM

---

[a] Measured at the stack.

TABLE 4. FLUE GAS SO_2 AND HCl CONCENTRATION MEASURED AT THE ESP INLET (ppmdv AT 7% O_2) AND PERCENT REDUCTION WITH LIME ADDITION

Set Point	SO2 (a)	SO2 (b)	HCl (c)	HCl (d)	Ca/(S+0.5*Cl) Mole Ratio	Percent Reduction SO2	Percent Reduction HCl
2A	127			445			
2B	183			473			
2C	158			620			
2D	110			499			
AVG(2A-2D)	144.5			509			
8B	73			293[e]	2.8	50	42
8A	35			384	1.87	76	25
	18	48			"	88-67	---
	25	37			"	83-75	---
8C	106	131	476[e]				
8AR	40		208		1.87	72	59
	58	49	221		"	60-66	57
	23	19	176		"	84-87	76
8BR	84				2.8	42	---
	106	115	333		0.0	20-26	34
	69	53	235		2.8	52-63	54
	67	39	172		"	54-73	66

---

[a] Continuous emission monitoring
[b] Manual sampling by EPA M6
[c] Manual sampling for HCl
[d] Manual sampling by HCl/Cr^{+6} sampling system
[e] Measured at the stack

TABLE 5. FLUE GAS TRACE METAL WEIGHT FRACTION MEASURED AT THE ESP INLET,
(wt % of total particulate)

	SET POINT						
ELEMENT	2A	2B	2C	2D	AVERAGE (2A-2D)	8B	8A
Arsenic	0.1110	0.0644	0.0712	0.0312	0.0694	0.0738	0.0675
Beryllium	<0.0010	<0.0012	<0.0017	<0.0014	<0.00133	<0.0013	<0.0013
Cadmium	0.0912	0.0761	0.1283	0.1457	0.11033	0.1245	0.0819
Chromium	0.2814	0.1175	0.0679	0.0808	0.1369	0.1076	0.0984
Lead	1.47	1.46	0.6470	0.9602	1.134	1.66	1.40
Mercury	0.0187	0.0466	0.1192	0.0465	0.0577	0.0554	0.0786

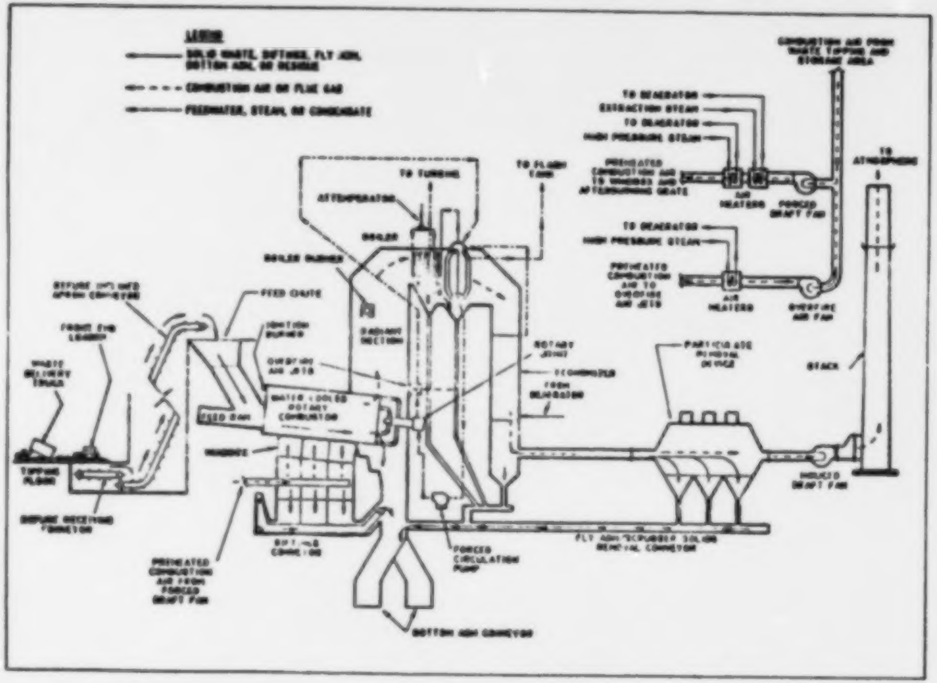
LEGEND
SOLID WASTE, SIFTINGS, FLY ASH, BOTTOM ASH, OR RESIDUE
COMBUSTION AIR OR FLUE GAS
FEEDWATER, STEAM, OR CONDENSATE
TO TURBINE
ATTEMPERATOR
BOILER
BOILER BURNER
TO FLASH TANK
RADIANT SECTION
FEED CHUTE
IGNITION BURNER
OVERFIRE AIR JETS
WATER COOLED ROTARY COMBUSTOR
FEED RAM
WINDBOX
REFUSE INCLINED APRON CONVEYOR
FRONT END LOADER
WASTE DELIVERY TRUCK
TIPPING FLOOR
REFUSE RECEIVING CONVEYOR
PREHEATED COMBUSTION AIR FROM FORCED DRAFT FAN
SIFTINGS CONVEYOR
BOTTOM ASH CONVEYOR
FORCED CIRCULATION PUMP
ROTARY JOINT
ECONOMIZER
FROM DEAERATOR
FLY ASH/SCRUBBER SOLIDS REMOVAL CONVEYOR
PARTICULATE REMOVAL DEVICE
INDUCED DRAFT FAN
STACK
TO ATMOSPHERE
COMBUSTION AIR FROM WASTE TIPPING AND STORAGE AREA
TO DEAERATOR
EXTRACTION STEAM
TO DEAERATOR
HIGH PRESSURE STEAM
PREHEATED COMBUSTION AIR TO WINDBOX AND AFTERBURNING GRATE
AIR HEATERS
FORCED DRAFT FAN
TO DEAERATOR
HIGH PRESSURE STEAM
PREHEATED COMBUSTION AIR TO OVERFIRE AIR JETS
AIR HEATERS
OVERFIRE AIR FAN

FIGURE 1. Bay County Plant Process Schematic Diagram

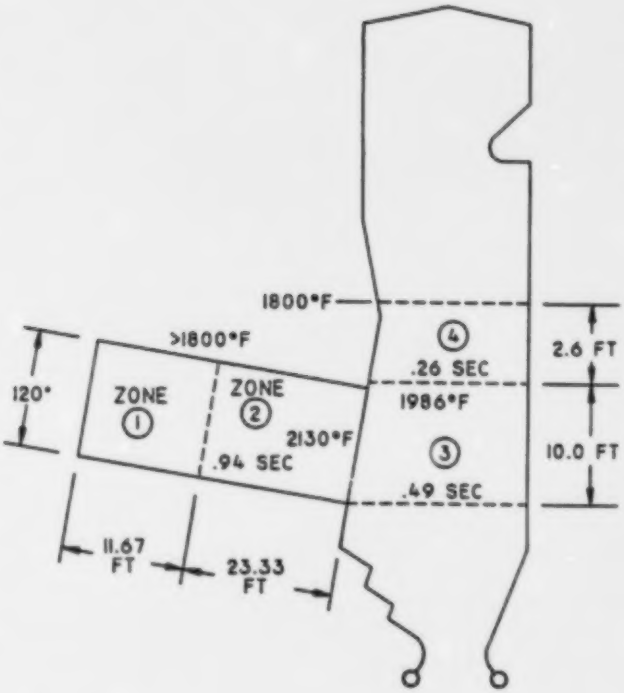
1800°F
>1800°F
120°
ZONE ①
ZONE ②
2130°F
.94 SEC
④
.26 SEC
1986°F
③
.49 SEC
2.6 FT
10.0 FT
11.67 FT
23.33 FT

FIGURE 2. Flue Gas Time/Temperature Profile

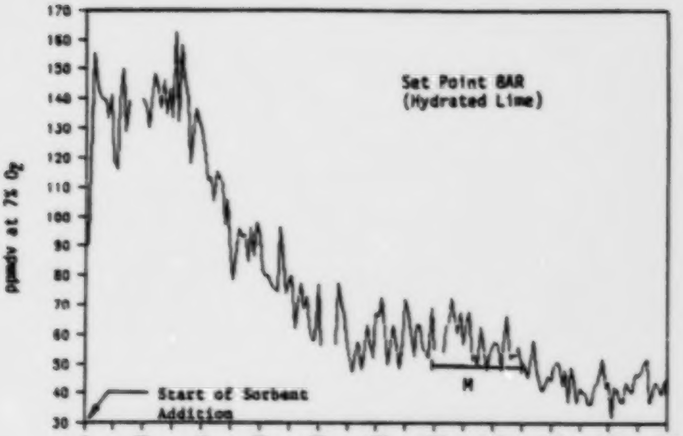

FIGURE 3. Flue Gas SO_2 Concentration

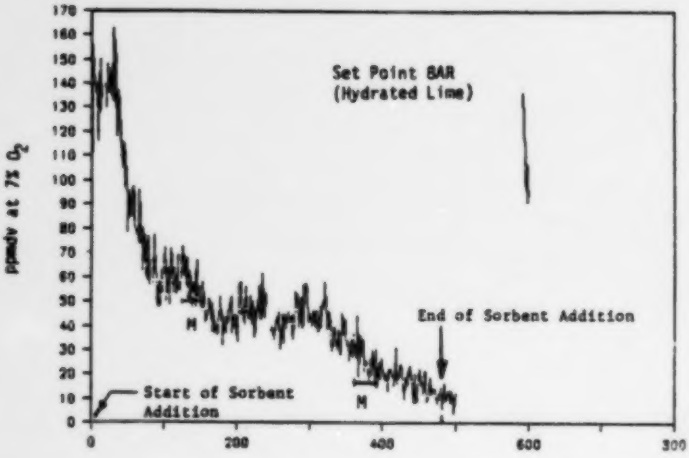

FIGURE 4. Flue Gas SO_2 Concentration

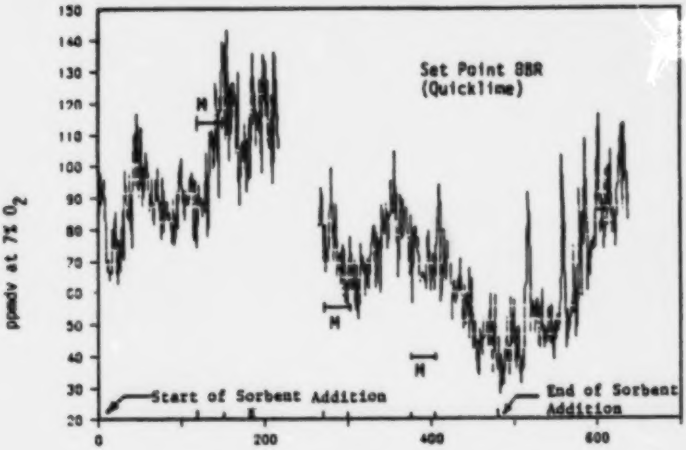

FIGURE 5. Flue Gas SO_2 Concentration

REMOVAL OF HYDROGEN CHLORIDE GAS FROM MSW FLUIDIZED BED INCINERATOR USING BAG FILTER

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ABSTRACT

An experiment in flue gas treatment using a bag filter was attempted by using actual gas from a municipal solid waste (MSW) fluidized bed incinerator, and the following results were obtained.

1) When the Ca/Cl equivalent ratio was 3, 80 to 95% of HCl could be removed.

2) The concentrations of polychlorinated dibenzo-p-dioxins (PCDD) and polychlorinated dibenzofurans (PCDF) were nearly same in the bag filter ash and EP (Electrostatic Precipitator) ash, but the concentrations of PCDD, PCDF in flue gas from bag filter were about 1/10 in flue gas from EP.

Therefore, when the bag filter system is incorporated in the MSW incineration plant, it is confirmed that removal of acid gas such as HCl and suppression of PCDD, PCDF are realized in addition to removal of dust.

INTRODUCTION

The amount of MSW in Japan is about 1.0 kg per person per day, and the total daily amount to 120 x 10^3 tons. About 70% of MSW is treated by incineration, and the remaining 30% is disposed by landfill. About 40 incinerators are built every year, and about 20% of them are the fluidized bed type. Recently, incineration plants are often built in suburban areas, and the flue gas regulations are becoming stricter every year. The emission standard in Japan is as shown in Table 1, but stricter standards has been setted by certain municipal governments, and we have been coping with such situations as shown in Table 1. The authors have been conducting experiments in flue gas treatment since 1986 by utilizing a bag filter test apparatus in an actual incineration plant. An outline of the results obtained so far are reported in this paper.

EXPERIMENT

APPARATUS AND PROCEDURES

The flow of the incineration plant and the bag filter test apparatus are shown in Figure 1. This plant is a fluidized bed incinerator for a medium-scale city, and has a capacity of 120 t/d (60 t/d x 2). The flue gas before dust removal was partially branched and introduced into the bag filter test apparatus, and after removal of dust, HCl, etc. in the bag filter, the gas was returned to the plant. Slaked lime powder [$Ca(OH)_2$] was injected into the upstream of the bag filter by a pneumatic conveyor through the powder feeder. For tests of high-concentration HCl, pure HCl gas was added. The characteristic of pressure control was carried out by both pressure difference control and regular interval pulse control. The specification of the bag filter test apparatus is shown in Table 2, and the test condition is given in Table 3.

MEASUREMENT AND ANALYSIS

The flue gas flow rate was measured by an orifice, and the temperature was measured by thermocouples. The amount of $Ca(OH)_2$ supplied was measured by using the calibration curve. The O_2 concentration was continuously measured by a magnetic wind system analyzer. The HCl concentration was continuously measured with an analyzer adopting the twin ion electrode method, and after 12% O_2 concentration conversion and moisture correction, the HCl removal efficiency was calculated from the following formula.

$$\text{HCl removal efficiency (\%)} = \frac{(\text{HCl})_{in} - (\text{HCl})_{out}}{(\text{HCl})_{in}} \times 100$$

The PCDD and PCDF were analyzed as follows. The ash analysis was conducted in the following manner. Exactly 20 g of dried ash sample was put in a cylindrical filter paper (glass fiber), and was extracted into 100 ml of toluene by using Soxhlet extractor for 6 hours. The obtained extract was then concentrated to about 10 ml, and further to 1 ml by nitrogen gas in a test tube, and was presented for the analysis. The flue gas was analyzed as follows. For measurement of the volatile content in flue gas, a 3-liter Tedlar bag was used as a sampling container, and for measurement of high boiling point components such as PCDD, a glass tube (6 mm dia. x 25 cm) was packed with 10 cm of Chromosorb WAW 60 to 80 mesh and 2 cm of activated carbon 30 to 60 mesh (cleaned in toluene and dried by using Soxhlet extractor), and both ends were plugged with glass wool, and this tube was heated to 300°C and aged with N_2 carrier to be used as a capturing tube. After sampling the gas, it was extracted by the Soxhlet extracted as in the case of ash, and the subsequent operations were exactly the same as with ash.

RESULTS

PRESSURE DROP OF BAG FILTER

To understand the pressure drop characteristic of the bag filter is an indispensable condition for practical use.

Generally, the pressure drop is expressed by the following formula.

$$\Delta P = \Delta P_f + \Delta P_d = (\zeta_f + m\alpha)\mu \cdot u/gc$$

where ΔP: pressure drop of bag filter [mmH_2O]
P_f: pressure drop of filter cloth [mmH_2O]
P_d: pressure drop of dust [mmH_2O]
ζ_f: resistance coefficient of filter [1/m]
m: dust load on filter [kg/m^2]
α: specific resistance of dust [m/kg]
μ: viscosity [kg/m·sec]
u: filtration velocity [m/sec]
gc: gravitational acceleration [$kg \cdot m/sec^2 \cdot kg$]

In the case of combustion exhaust gas, ζf, μ and α are supposed to be constant, the pressure drop may be expressed as a function mainly of the filtration velocity (u) and dust load on filter (m). Figure 2 shows an example of measurement of pressure drop (ΔP) and filtration velocity (u). From this figure, the following relation is obtained.

$$\Delta P = u^{1.1}$$

Though the dust load on filter could not be measured, we could estimate that the thickness of ash deposit was about 4 mm at m_1, and about 15 mm at m_3. The control of pressure drop was tested in both the regular interval pulse method, and the pressure difference method for pulsating between upper and lower limits. In both methods, the control of pressure drop was excellent, and one example of the pressure drop control by the pressure difference method is shown in Figure 3. However, since the moisture content in flue gas is as high as 30 to 35 vol.%, and reaction products such as $CaCl_2$ which has moisture absorbing property are contained in the dust, therefore caution is needed when stopping and starting the incineration plant in particular.

HCl REMOVAL

By injecting powder of $Ca(OH)_2$ into upstream of the bag filter, it was tried to remove HCl in the duct and on the surface layer of the filter cloth. The reaction formula is as follows.

$$2HCl + Ca(OH)_2 \rightarrow CaCl_2 + 2H_2O$$

Figure 4 shows the effects of filtration velocity. As shown in the figure, in a range of filtration velocity of 0.8 to 1.2 m/min, there is hardly any effect on the HCl removal efficiency. The temperature effects are shown in Figure 5. In an ordinary operating temperature range of 170 to

230°C, the effect of the temperature on the HCl removal efficiency is slight, and in this region it is estimated that diffusion rather than reaction, especially the diffusion within a particle is the rate-determining step.

In Figure 6, the relation between the Ca/Cl equivalent ratio and the outlet HCl concentration is given. As can be seen from the figure, as the Ca/Cl equivalent ratio becomes larger, the outlet HCl concentration becomes nearly constant regardless of the inlet HCl concentration. This is because, even at the same equivalent ratio, when the inlet HCl concentration is higher, the $Ca(OH)_2$ amount per unit flue gas volume increases and the solid gas contact efficiency increases and also the $Ca(OH)_2$ inventry in the dust deposit on the filter increases. Thus, by injecting about 3 equivalents of $Ca(OH)_2$ into upstream of the bag filter, an HCl removal efficiency of about 80 to 95% could be obtained, and the outlet HCl concentration could be lowered to 50 ppm or less. When the expected value is less than 50 ppm, the wet scrubbing process was generally applied, but it has been found that, by using the bag filter, the flue gas reheating equipment for prevention of visible plume and scrubbing water treatment are not necessary, and these are major advantages.

SUPPRESSION OF DIOXIN FORMATION

In order to understand the behavior of PCDD and PCDF in the incineration plant, the concentrations were measured at the following positions.

[Gas]	
1. Cooling tower outlet	(G-1)
2. EP outlet	(G-2)
3. Bag filter outlet	(G-3)
[Ash]	
1. Cooling tower bottom	(A-1)
2. EP bottom	(A-2)
3. Bag filter bottom	(A-3)

An example of measured data is shown in Table 4, from which the following results were found.

1) PCDD and PCDF are hardly detected in both gas and ash at the cooling tower outlet.

2) The concentrations of PCDD, PCDF in flue gas from bag filter are less than 1/10 in flue gas from EP.

3) The concentrations of PCDD, PCDF in ash are not significantly different between the bag bottom ash and EP bottom ash.

Comparing PCDD and PCDF in the ash with ones in the gas, in the case of EP, 70 to 90% is discharged from the gas side. And the ratio of adsorption on ash tended to be higher when the number of chlorine was increased, because, these vapor pressure was lower. Meanwhile, chlorinated benzene which is higher in vapor pressure than PCDD, PCDF is contained at the gas side by more than 99%. Estimating from the above findings, we can say that PCDD, PCDF are considered to be formed in the cooling process, especially in the EP, rather than being produced in the furnace.

The following results were confirmed by Hanai, et al.,(1) in the ash heating test and the corona discharge test.

1) The chlorination reaction can be notably suppressed when the operating temperature is controlled under 200°C.

2) The chlorination reaction can be suppressed when the ash is not exposed to high temperature condition for a long time.

3) Increasing the corona discharge current, the production of chlorine compounds increase. (By the corona discharge, the HCl is dissociated to produce highly reactive Cl, Cl_2, thereby the chlorination reaction is promoted.)

These results suggest that the operating conditions of temperature, residence time, corona discharge at EP are suitable for encouraging the production of PCDD, PCDF. Figure 7 shows the test results at different operating temperatures of the bag filter, in comparison with the EP data. The EP was usually operated at 300°C. The high number chlorine PCDD can be found out in flue gas from EP than that of bag filter, so we can see that the chlorination is further advanced in EP.

The relation between the total discharge ratio and gas temperature is shown in Figure 8. As shown in the figure, by controlling the operating temperature at 200°C, the total discharge of PCDD, PCDF in the gas and ash can be suppressed to about 1/10 as compared with the EP. The mechanism of formation of chlorinated compounds in EP is supposed as follows. Generally in the incinerator, since the temperature is as high as 750 to 850°C, chlorination reactions hardly take place, and if chlorinated compounds are produced, they are immediately oxidized. On the other hand, the EP temperature is 300°C, and the residence time is long, so that the chlorination reaction is promoted without being affected by the oxidation reaction. The compounds originating from this process are macromolecular organic components left over without vaporized from the ash, and they are thought to operate to produce chlorinated compounds aided by metal compounds in the ash, such as $FeCl_3$, as the catalyst. Therefore, when the bag filter system is installed instead of the EP, it is found that the gas side discharge of the PCDD and PCDF can be reduced.

Vogg. et al.,(2)(3) reported that PCDD, PCDF in ash could be decomposed by heating it over about 500°C. Authors have also got the same results through the continuous heating test of ash.

CONCLUSION

In Japan EP has been used widely for the incineration plant of MSW, but from the results of the present test, it has been found that the bag filter system cannot only remove the HCl efficiently, but also reduce the concentrations of PCDD, PCDF in the gas. Meanwhile, the PCDD and PCDF in ash are known to be decomposed by reheating the ash, and further data will be accumulated henceforth. The bag filter system can also remove SOx, and may catch heavy metals from flue gas because the operating temperature can be lowered. In the future, therefore, it is expected that the application of bag filters in the field of incineration of MSW will increase furthermore. At the same time, recently, the development of new filter cloths is intensive, then this will also help to spread bag filters in a wider field.

REFERENCES

(1) Hanai, Y., Kato, T., Ide, H. Chlorination process in the refuse incinerator's equipments. Bulletin of the Institute of Environmental Science and Technology of Yokohama National Univ. 13. 1986.
(2) Vogg, H., Stieglitz, L. Thermal behavior of PCDD/PCDF in fly ash from municipal incinerators, Chemosphere, 15, 1986.
(3) Vogg, H., Stieglitz, L. On formation condition of PCDD/PCDF in fly ash from municipal incinerators, chemosphere, 16, 1987.

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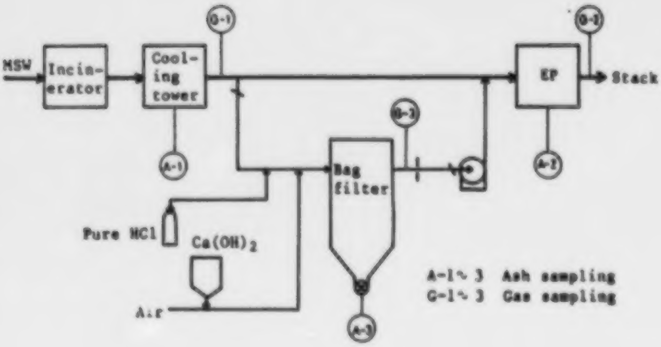
MSW
Incin-erator
Cool-ing tower
G-1
EP
Stack
G-2
G-3
A-1
A-2
A-3
Bag filter
Pure HCl
Ca(OH)₂
Air
A-1∼3 Ash sampling
G-1∼3 Gas sampling

Figure 1. Schematic of Bag Filter Test System

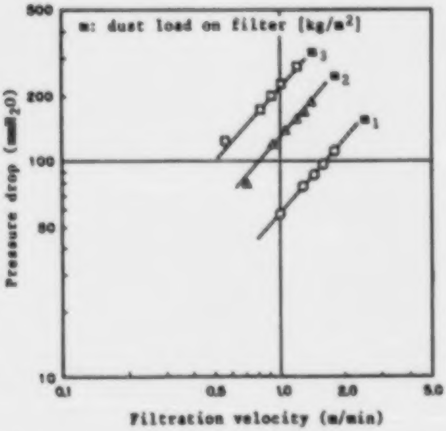

Figure 2. Relations between Filtration Velocity and Pressure Drop

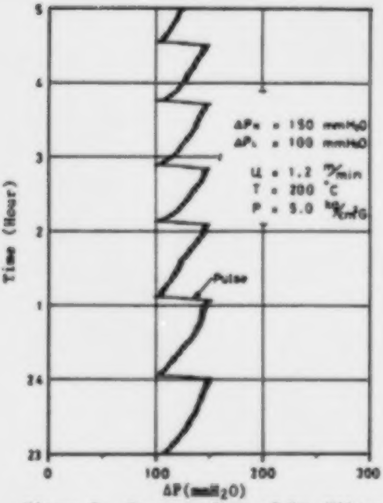

Figure 3. Pressure Drop of Bag Filter

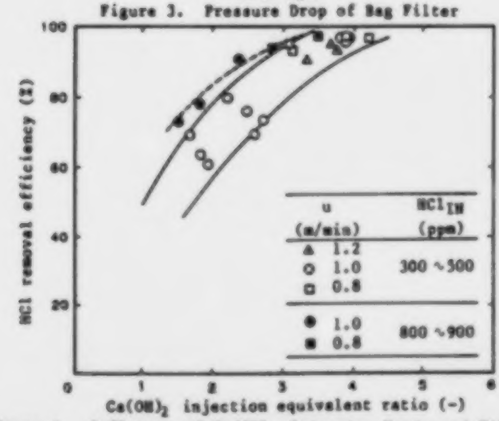

Figure 4. Influence of $Ca(OH)_2$ Injection Ratio and Filtration Velocity on HCl Removal Efficiency

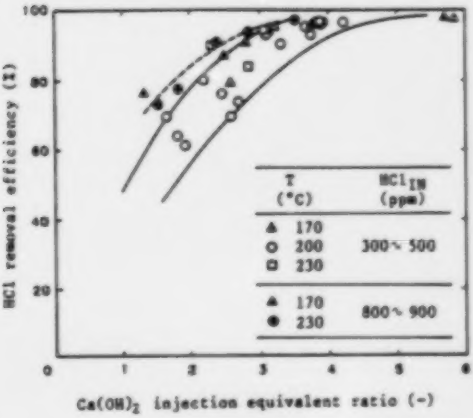

Figure 5. Influence of Ca(OH)$_2$ Injection Equivalent Ratio and Gas Temperature on HCl Removal Efficiency

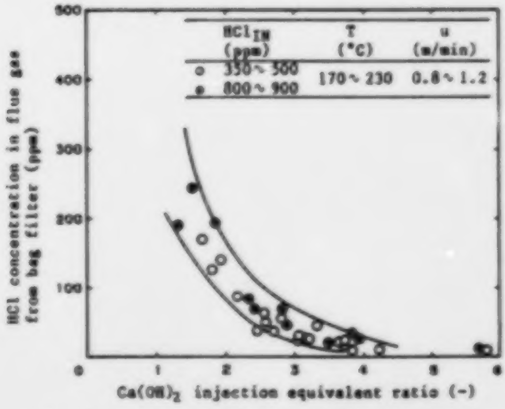

Figure 6. Influence of Ca(OH)$_2$ Injection Equivalent Ratio on HCl Concentration in Flue Gas from Bag Filter

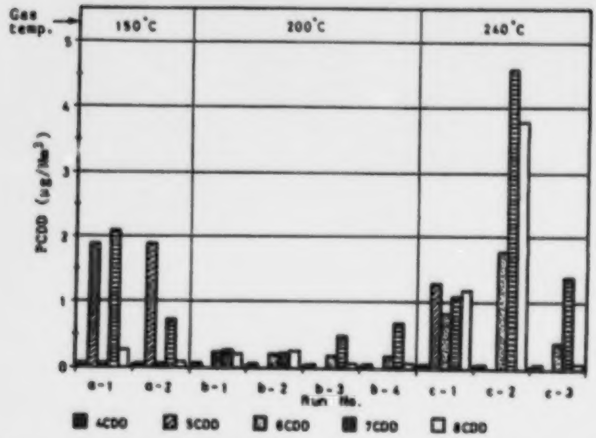

Figure 7-1. PCDD Homologue Distribution in Flue Gas from Bag Filter

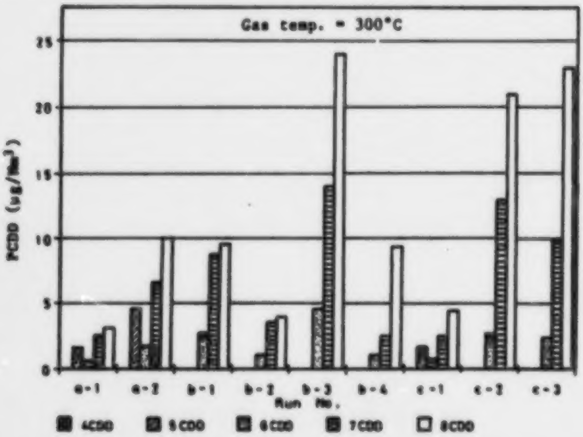

Figure 7-2. PCDD Homologue Distribution in Flue Gas from EP

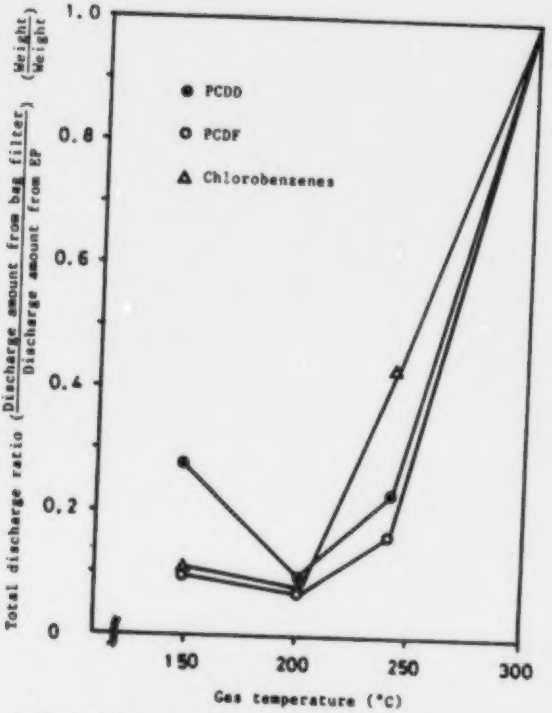

Figure 8. Relations between Gas Temperature and Total Discharge Ratio

TABLE 1. POLLUTION CONTROL

	Emission standard	Counter-measure	Expected figure
Dust (mg/Nm^3)	80	None	10,000 - 15,000
		EP	< 30
NOx (ppm)	250	FBI	<150
		2-stage combustion	<100
		Catalyst	< 20
HCl (ppm)	430	None	300 - 800
		$CaCo_3$ into incinerator	<200
		$Ca(OH)_2$ into EP inlet	<100
		Wet scrubbing	< 30
SOx (ppm)	160 (in Tokyo)	None	10 - 40
		Wet scrubbing	< 20

TABLE 2. SPECIFICATION OF BAG FILTER

Filter size and number	116 ϕm/m x 1300^L m/m x 9
Area of filtration	4.2 m^2
Filter materials	Glass felt, Teflon laminated glass cloth, Woven glass, etc.
Filter cleaning method	Pulse jet

TABLE 3. EXPERIMENTAL CONDITION

Wet gas volume	120 ~ 180 Nm^3/h
Gas temperature	170 ~ 230°C
Inlet HCl	300 ~ 500 ppm 800 ~ 900 ppm (pure HCl inject)
Filtration velocity	0.8 ~ 1.2 m/min
Pulse pressure	5.0 kg/cm^2G (4.9×10^5 Pa)

TABLE 4. CONCENTRATIONS OF CHLORINATED COMPOUNDS IN ASH AND FLUE GAS FROM THE MSW INCINERATION PLANT

	Ash (ng/g)			Flue gas ($\mu g/m^3$)		
	Cooling tower (A-1)	EP (A-2)	Bag filter (A-3)	Cooling tower outlet (G-1)	EP outlet (G-2)	Bag filter outlet (G-3)
PCDD						
4CDD	nd	nd	nd	nd	nd	nd
5CDD	nd	nd	nd	nd	nd	nd
6CDD	nd	18.0	24.0	nd	1.2	0.2
7CDD	nd	130.0	74.0	nd	2.6	0.7
8CDD	nd	270.0	80.0	nd	9.4	nd
Total PCDD	0.0	418.0	178.0	0.0	13.2	0.9
PCDF						
4CDF	nd	12.0	15.0	nd	11.0	1.2
5CDF	nd	34.0	42.0	0.44	16.0	1.8
6CDF	nd	140.0	92.0	nd	36.0	3.8
7CDF	2.3	120.0	43.0	0.50	11.0	nd
8CDF	nd	180.0	76.0	nd	19.0	nd
Total PCDF	2.3	486.0	268.0	0.94	93.0	6.8
Crolobenzens						
3-	0.9	60.0	110.0	3.9	960	6.4
4-	7.2	83.0	140.0	9.1	1270	25.0
5-	8.9	210.0	260.0	15.0	3160	24.0
6-	5.2	150.0	240.0	2.9	350	6.7
Total CBs	22.2	503.0	750.0	30.9	5740	62.1

Pilot Evaluation of Sorbents for Simultaneous Removal of HCl and SO_2 from MSW Incinerator Flue Gas by Dry Injection Process

by

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Hollywood, FL, April 11-14, 1989

ABSTRACT

This paper presents the results of pilot plant tests that evaluated the potential sorbents for use in a dry-injection process designed to control HCl and SO_2 flue gas emission. Baseline data were developed with high calcium hydrated lime. Particular attention was given to difficulties associated with operating at low temperatures owing to the presence of $CaCl_2$. Magnesium and sodium based sorbents were then evaluated for direct comparison to lime in terms of sorbent cost for acid gas emission control.

A key finding is the degree to which the flue gas temperature can be lowered with the dry-injection process while maintaining operating reliability by eliminating the $CaCl_2$ drying problem associated with wet wall deposits. Although not measured, lower temperature operation also has significant potential for lowering emissions of volatile organics (dioxins) and trace metals. The interaction between HCl and SO_2, removed using calcium hydroxide, can be significantly increased or decreased through interactions in solid-phase mass-transfer rates.

This paper has been reviewed in accordance with the U.S.
Environmental Protection Agency's peer and administrative
review policies and approved for presentation and publication.

INTRODUCTION

In-duct dry-sorbent injection technology [1] involves injecting a dry sorbent in conjunction with flue gas humidification by water spraying in the ductwork between the air preheater and the particulate collector (electrostatic precipitator or baghouse) in a municipal solid waste (MSW) incinerator system.

With a humid flue gas, acid gases such as HCl and SO_2 are removed by the dispersed sorbent particles in the duct and by the sorbent holdup in the particulate collector. Figure 1 is a schematic diagram of the in-duct dry sorbent injection process. The flue gas coming out of the air preheater is humidified by atomized water sprays to a close approach to adiabatic saturation temperature. The dry sorbent, usually a fine powder, is injected into the duct, either upstream or downstream of the humidification zone. When the sorbent is injected upstream of the humidification zone, some interaction may occur between the sorbent particles and the atomized water [2]. The mechanism of this interaction may be the wetting of sorbent particles through direct droplet-particle collisions. The water content in the wetted sorbent particles could reach a level well above that possible by physical adsorption alone. Sorbent reactivity toward SO_2 could be increased through this mechanism. In the presence of HCl, however, drying difficulties may be encountered due to the strong water retaining properties of absorbed HCl and dissolved chloride salt. The wet slow-drying product has a potential of causing wet wall deposits and operating problems with the particulate collection system. On the other hand, most of the sorbent particles/evaporating droplets interactions can be avoided with the injection of sorbent downstream of the water sprays. Since the sorbent is added after most of the sprayed water droplets have evaporated, the surface moisture of the particles can be increased only by physical adsorption/condensation. Only several layers of water molecules are expected to form through this mechanism according to equilibrium data [3]. The potential for wet-wall deposits and a sticky sorbent bed in particulate collectors is at a minimum. Therefore, sorbent injection downstream of the humidification zone is the process scheme used for current tests.

Compared with the traditional means of acid gas removal, spray drying followed by a baghouse, the in-duct dry-sorbent injection process offers a technically simple and low cost alternative. The main savings are derived from eliminating the spray drying vessel, which has both high initial and high maintenance costs, and slurry atomization nozzles. In addition, if the existing ductwork and particulate collector are used, the in-duct dry-injection process provides low-cost retrofit acid gas control options for existing MSW incinerators.

Because of the drying problems associated with wet chloride salts, spray dryers have usually been operated at a very high temperature [greater than 60 °C (108 °F) approach to saturation]. But when the sorbent is injected dry and downstream of the humidifier, the temperature can be reduced to below 40 °C (72 °F) approach to saturation, which favors the removal of volatile organic compounds and heavy metals.

To evaluate and develop dry-injection acid gas removal, Jozewicz et al. conducted bench-scale experiments to assess commercially available sorbents that can potentially be used for dry-injection purposes. Among them, hydrated lime, magnesium hydroxide, and sodium bicarbonate are the most feasible. In the present study, small-scale pilot plant tests were calculated to further evaluate the performance of these three sorbents. This paper summarizes the pilot plant tests and discusses implications for large-scale and commercial applications. Data are also presented to illustrate the interaction of HCl and SO_2 absorption mechanism by the most promising sorbent.

TEST FACILITY

PILOT PLANT

Figure 2 is a schematic diagram of the pilot plant, which includes a simulated flue gas generation system, the humidifier, the sorbent injection system, the ductwork, a baghouse, and a fan. Air is drawn through a 3-phase electrical heater that raises the gas temperature to about 204 °C (400 °F). Pure SO_2 and HCl are injected into the heated gas stream to simulate incinerator flue gas conditions. The pilot humidifier uses an air-driven rotary atomizer to spray water into the hot air. The sorbent injection system allows for the metered addition of solid powders into the gas stream at a point about 2 m downstream of the humidifier exit. The system uses a volumetric solids feeder mounted on an electronic scale with casters to allow quick calibration and easy placement in and out of the pilot plant. The sorbent particles are transported by the volumetric feeder helix to the feed port; they then drop into a vertical section of the duct. No gas/solid mixing devices are used to disperse the injected solids. The stainless-steel duct connecting the humidifier and the baghouse is heat traced and insulated to maintain the desired gas temperature throughout the system.

The duct section between the humidifier and the baghouse inlet is 24.4 m (80 ft) long and has a 5 cm (2 in.) O.D. with a gas residence time of about 2 s. The solid product is removed from the gas stream inside a pulse jet baghouse. Four Teflon bags, each 15 cm (6 in.) in diameter and 1.2 m (48 in.) long, provide an air-to-cloth ratio of about 1 cm/s (2 scfm/ft^2). The entire baghouse is insulated and enclosed in a wooden cabinet. An induced draft fan, downstream of the baghouse, draws the simulated flue gas through the pilot plant. The gas flow rate is monitored and controlled by an orifice plate and a butterfly valve. Normally, the air flow rate is controlled at 85 m^3/h (50 scfm).

GAS SAMPLING AND ANALYSIS

Sample ports for SO_2 are available at the humidifier outlet (before the sorbent injection port), baghouse inlet, and baghouse outlet. Particulate entrainment is a concern at the baghouse inlet sampling port. Solids are removed from the sample gas with minimum interaction between the gas and the sorbent solids. The sample gas is extracted from the duct just before the baghouse inlet through an inverted flared nozzle that diverts solid particles around the intake. The solids remaining in the sample gas are removed in a particulate filter heated to above 60 °C (108 °F) approach to saturation to prevent continued SO_2 absorption by the filtered particles. Heat traced Teflon lines are used to transport the sample gas to a pulse-fluorescent, continuous emission monitor for SO_2 analysis.

Two sample ports, one at the humidifier outlet and one at the baghouse outlet, are available for HCl sampling. The HCl sampling system includes valves, fittings, and tubing and is heated to 150 °C (302 °F). The sample line is connected to a glass impinger train that consists of four impingers partially immersed in an ice bath. Deionized water is added to the first two impingers for HCl collection. A vacuum-leak check is conducted prior to each sampling, performed at a ratio of 1 L/min (0.5 ft^3/h) for 30 min. After each sample, the heated sample line is purged with air for 25 min. The wet sample recovered from the impinger train is analyzed by ion chromatography for chloride concentration.

RESULTS AND DISCUSSIONS

SORBENT EVALUATION TESTS

Three potential sorbents, hydrated lime, sodium bicarbonate, and magnesium hydroxide were evaluated in the pilot plant. The flue gas SO_2 and HCl concentrations were controlled at 200 and 600 ppm, respectively, and were measured at the humidifier outlet and baghouse outlet to determine the removal efficiencies. Approach to saturation temperature and sorbent stoichiometric ratio are the major independent variables. Stoichiometric ratio is important because it relates the amount of injected sorbent to the total amount of acid gases (HCl and SO_2) present in the inlet flue gas. It affects the operating cost of dry injection acid gas removal process.

When hydrated lime is used as the sorbent, the reaction is assumed to be:

$$Ca(OH)_2 + SO_2 \xrightarrow{\hspace{2cm}} CaSO_3 + H_2O$$

$Ca(OH)_2 + 2HCl \longrightarrow CaCl_2 + 2H_2O$

Accordingly, stoichiometric ratio is defined as:

$$\text{Stoichiometric ratio for } Ca(OH)_2 = \frac{\text{moles of } Ca(OH)_2}{1/2 \text{ moles HCl} + \text{moles of } SO_2}$$

Figure 3 shows SO_2 and HCl removal efficiencies plotted against approach to saturation temperature using stoichiometric ratio as a parameter. Over 99 percent HCl removal efficiency was obtained for all the tests using hydrated lime as the sorbent. In other words, HCl removal efficiency was not sensitive to temperature and stoichiometric ratio within the operating range used for dry-injection pilot plant testing. On the other hand, SO_2 removal efficiency was very sensitive to both operating parameters: it increased significantly as stoichiometric ratio increased and approach to saturation temperature decreased. More than 90 percent SO_2 was removed when the approach to saturation temperature was lower than 28 °C (50 °F) and when the stoichiometric ratio was greater than 1.0. Lowering the approach to saturation to 22 °C (40 °F) resulted in nearly 100 percent SO_2 removal efficiency. Operating difficulties were encountered, however, when the approach to saturation temperature was below 22 °C (40 °F). Wet deposits were found on the duct wall downstream of the sorbent feed point. The pressure drop across the ductwork increased significantly, and long-term operation was not sustainable.

The second sorbent tested was sodium bicarbonate. The reaction is assumed to be:

$2NaHCO_3 + SO_2 \longrightarrow Na_2SO_3 + H_2O + 2CO_2$

$NaHCO_3 + HCl \longrightarrow NaCl + H_2O + CO_2$

Sorbent stoichiometric ratio is defined as:

$$\text{Stoichiometric ratio for } NaHCO_3 = \frac{\text{moles of } NaHCO_3}{\text{moles of HCl} + 2 \text{ moles } SO_2}$$

Figure 4 shows the performance curves of sodium bicarbonate. The stoichiometric ratio and temperature had little effect on HCl removal efficiency. Nearly 100 percent HCl removal was obtained from all the tests. For SO_2 removal, stoichiometric ratio and approach to saturation temperature had significant effects. In general, the SO_2 removal increased when the stoichiometric ratio increased. At stoichiometric ratio of 1.0, the SO_2 removal efficiency decreased with the decrease of approach to saturation, which is just the opposite of the effect observed for hydrated lime. Operating difficulties with wet-wall deposits and high system-pressure drop were encountered when the approach to saturation temperature was lowered to 22 °C (40 °F) or below.

The third sorbent was magnesium hydroxide. The reaction is assumed to be:

$Mg(OH)_2 + SO_2 \longrightarrow MgSO_3 + H_2O$

$Mg(OH)_2 + 2HCl \longrightarrow MgCl_2 + 2H_2O$

Stoichiometric ratio is defined as:

$$\text{Stoichiometric ratio for } Mg(OH)_2 = \frac{\text{moles of } Mg(OH)_2}{1/2 \text{ moles } HCl + \text{moles of } SO_2}$$

Figure 5 shows that, like for the other sorbents tested, significant HCl removal efficiency was obtained from the tests. The removal efficiency for SO_2, however, was considerably lower than for HCl at stoichiometric ratio of 1.0. With $Mg(OH)_2$, stoichiometric ratio and approach to saturation temperature affected SO_2 removal efficiency in a manner similar to that with hydrated lime. However, at equivalent stoichiometric ratio and approach to saturation temperature, the SO_2 removed with $Mg(OH)_2$ is slightly lower than with $Ca(OH)_2$. Operating difficulties similar to those encountered with hydrated lime and sodium bicarbonate occurred when the approach to saturation temperature was below 22 °C (40 °F).

Using the pilot plant data presented in Figures 3-5, sorbent cost was analyzed. Table 1 compares the cost of the three sorbents tested at two temperatures. The sorbent stoichiometric ratio required to remove 90 percent SO_2 and 99 percent HCl was obtained by interpolating the data shown in Figures 3-5. Table 1 shows that hydrated lime is the most cost-effective sorbent of the three tested.

$Ca(OH)_2$ MASS TRANSFER TESTS

Figure 3 shows that a significant amount of SO_2 was removed with hydrated lime as the sorbent when the pilot plant was operated at a stoichiometry of 3.0 and the approach to saturation was controlled below 33 °C (60 °F). However, when the pilot plant was operated at the same conditions (same inlet SO_2 concentration, lime feed rate, approach to saturation, etc.) but without HCl, considerably less SO_2 was removed. Figure 6 shows that, at 28 °C (50 °F) approach to saturation and 200 ppm inlet SO_2 concentration, only 65 percent SO_2 was removed when no HCl was present in the simulated flue gas. Figure 3 shows that about 97 percent SO_2 was removed when 600 ppm HCl was present in the inlet flue gas. The presence of 600 ppm HCl significantly enhanced SO_2 removal efficiency. A series of mass transfer tests were therefore conducted to investigate the possible synergistic effect between SO_2 and HCl absorption by dry-injected lime.

The results are shown in Figure 7, which illustrates the effect of HCl addition on SO_2 removal at 28 °C (50 °F) approach to saturation. The figure can be divided into three regions according the inlet HCl concentrations. When the concentration is less than 450 ppm, there is a positive relationship between SO_2 removal enhancement and HCl addition rate. This enhancement effect peaked at about 450 ppm inlet HCl concentration. The second region is between 450 and 750 ppm inlet HCl concentration.

TABLE 1. COMPARISON OF SORBENT COST PER KILOGRAM OF SO_2

Sorbent	Price \$/kg[a]	Approach to Saturation °C (°F)	SO_2 Removal %	HCl Removal %	Stoichiometric Ratio	Sorbent Cost \$/kg of SO_2
Hydrated lime	0.05	28 (50)	90	99	2.0	0.12
$NaHCO_3$	0.39	28 (50)	90	99	1.8	0.92
$Mg(OH)_2$	1.71	28 (50)	90	99	2.5	3.87
Hydrated lime	0.05	33 (60)	90	99	2.8	0.16
$NaHCO_3$	0.39	33 (60)	90	99	1.5	0.77
$Mg(OH)_2$	1.71	33 (60)	90	99	3.0	4.65

[a] Based on December 26, 1988, issue of Chemical Marketing Report.

Although SO_2 removal is still greater in this region than without HCl addition, the relationship between SO_2 removal and inlet HCl concentration becomes negative. In the third region, where the inlet HCl concentration increases beyond 750 ppm, SO_2 removal efficiency was even below that when no HCl was present in the inlet flue gas.

Figure 7 suggests strong interactions between HCl and SO_2 absorption mechanisms. Before further analysis of the simultaneous HCl/SO_2 absorption data, the baseline SO_2 absorption mechanism (i.e., SO_2 absorption in the absence of HCl) should be examined. The baseline SO_2 removal efficiency data shown in Figure 6 can be transformed and replotted as SO_2 absorption rate represented by open squares in Figure 8. The baseline SO_2 absorption rate is controlled largely by solid-phase mass transfer, especially when the inlet SO_2 concentration is greater than 200 ppm.

This control mechanism is supported by the evidence that the baseline SO_2 absorption rate remained constant when the inlet SO_2 concentration increased from 300 to 950 ppm. That is, the baseline SO_2 absorption was not affected by significant gas-phase concentration changes. However, when the lime feed rate was increased from 5.7 to 7.7 gmol/h, the baseline SO_2 absorption rate increased almost proportionally. In other words, the baseline SO_2 absorption rate is strongly affected by the solid-phase sorbent feed rate, which also indicates that the SO_2 absorption mechanism is solid-phase mass-transfer controlled.

Therefore, one conclusion that can be drawn from the comparison of the baseline SO_2 absorption rate (Figures 6 and 8) and the SO_2 absorption rate in the presence of HCl (Figure 7) is that the interaction of SO_2/HCl absorption is through a solid-phase mass-transfer mechanism.

This conclusion is also supported by Figure 3, which shows that nearly 100 percent HCl was removed achieved when SO_2 removal was low. The suggested mechanism for simultaneous HCl/SO_2 removal is that the more reactive and acidic HCl is absorbed by injected lime in advance of SO_2. The absorbed HCl changes the surface chemical/physical conditions of the lime particles through which the solid-phase mass transfer of SO_2 removal is affected. The absorbed HCl can affect SO_2 removal either positively or negatively, as shown in Figure 7. If $CaCl_2$ is formed on the surface of lime particles, the hygroscopic property of $CaCl_2$ would greatly increase the surface water content, thereby enhancing SO_2 removal efficiency. On the other hand, the absorption of HCl neutralizes and consumes reactive calcium hydroxide and forms chemically inert calcium chloride, which tends to hinder solid-phase SO_2 mass transfer. As Figure 7 indicates, the positive effects of HCl on SO_2 absorption dominate when the inlet HCl concentration is below 450 ppm. When the inlet HCl concentration exceeds 750 ppm, the initial HCl absorption consumes a significant amount of sorbent and causes considerable mass-transfer resistance against the subsequent SO_2 mass transfer. More mass-transfer data are needed to further analyze the simultaneous HCl/SO_2 absorption mechanism and to quantify the positive/negative interaction of the two acid gas absorption rates.

An interesting phenomenon observed during the mass-transfer testing is the generation of SO_2 by HCl injection. In one case, only HCl was injected into the simulated flue gas. Instead of remaining at zero, the SO_2 concentration at the baghouse outlet increased to about 300 ppm. In another case, the sorbent feeder malfunctioned in the middle of a test and lime injection was interrupted; the baghouse outlet SO_2 concentration drifted upward and exceeded the inlet SO_2 concentration. In other words, both cases indicate that SO_2 was produced and that the system outlet SO_2 concentration was higher than that at the inlet. It is believed that the SO_2 generation was caused by the replacement reaction:

$$2HCl + CaSO_3 \longrightarrow CaCl_2 + H_2O + SO_2$$

Because HCl is more reactive and acidic than SO_2, the $CaSO_3$ formed (by SO_2 absorption) and accumulated in the baghouse (as filter cake on the bags) is subject to HCl attack. When a significant amount of HCl is present in the baghouse and when there is a deficiency of active sorbents such as hydrated lime, the replacement reaction occurs and SO_2 is released from the solids collected in the baghouse. It should be noted that the SO_2 generation phenomenon is not unique to the dry injection process. The same replacement reaction is expected to occur in spray drying if there is a deficiency of sorbent (owing to poor slaking or equipment malfunction) or a sudden increase of HCl concentration.

SUMMARY

Three potential sorbents, hydrated lime, sodium bicarbonate, and magnesium hydroxide, were evaluated by pilot plant tests for simultaneous HCl/SO_2 removal. Hydrated lime was found to be the most cost-effective sorbent among the three tested for in-duct, dry-injection, acid gas removal. The pilot plant results show that HCl removal should not be a concern under the in-duct, dry-injection conditions tested because greater than 99 percent HCl was removed. When hydrated lime was used as the sorbent, SO_2 removal was favored by high stoichiometric ratio and low approach to saturation. More than 90 percent SO_2 was removed at stoichiometric ratios less than 3 and approach to saturation temperatures below 33 °C (60 °F). Approach to saturation of 22 °C (40 °F) represents the operating limit of in-duct dry-injection operation for simultaneous HCl/SO_2 removal because wet-wall deposits and high system-pressure drop were encountered in the pilot plant.

Mass transfer data show that the presence of HCl can significantly affect SO_2 absorption rate by hydrated lime. The solid-phase mass transfer of SO_2 in $Ca(OH)_2$ particles was enhanced by the addition of HCl in the flue gas. The enhancement effect increased to a peak with the increase of HCl addition rate and then diminished. A negative effect for HCl addition on SO_2 removal was obtained when the HCl concentration was greater than 750 ppm. Thus, with an excess of HCl and a deficiency of calcium hydroxide, the HCl may replace absorbed SO_2, and gaseous SO_2 may evolve from the baghouse. This phenomenon would be expected only when there was a sudden change in flue gas conditions or a malfunction of major equipment.

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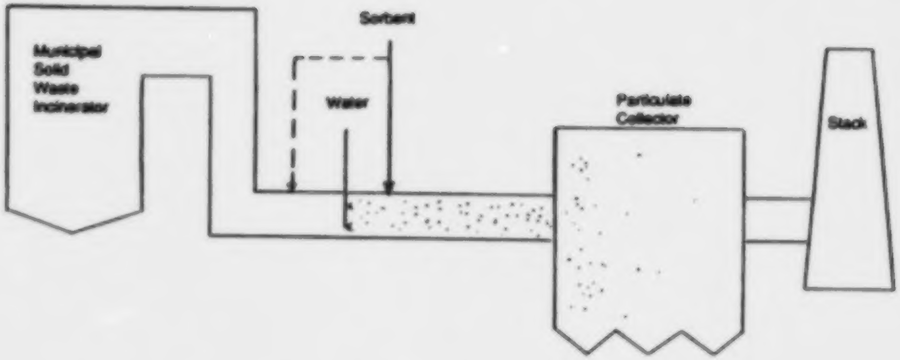
Municipal
Solid
Waste
Incinerator
Sorbent
Water
Particulate
Collector
Stack

Figure 1. Schematic diagram of in-duct dry-injection acid gas removal process

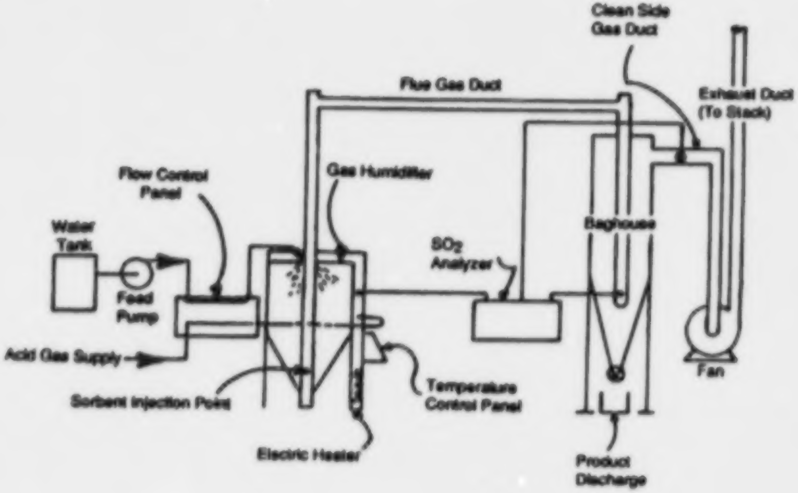
Clean Side Gas Duct
Flue Gas Duct
Exhaust Duct (To Stack)
Flow Control Panel
Gas Humidifier
Water Tank
Feed Pump
Baghouse
SO_2 Analyzer
Acid Gas Supply
Fan
Sorbent Injection Ports
Temperature Control Panel
Electric Heater
Product Discharge

Figure 2. Schematic of duct-injection/pilot plant facility.

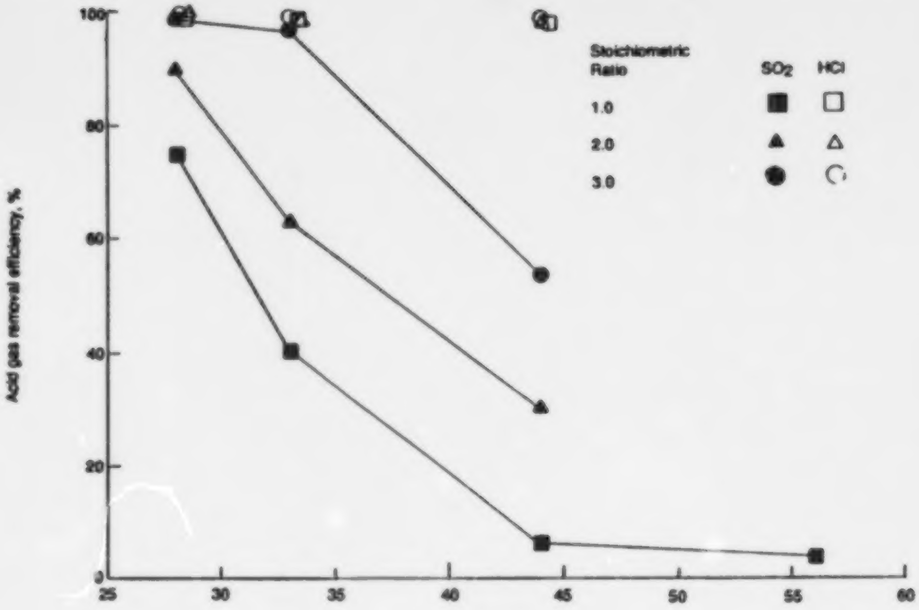

Figure 3. Acid gas absorption rates of hydrated lime.

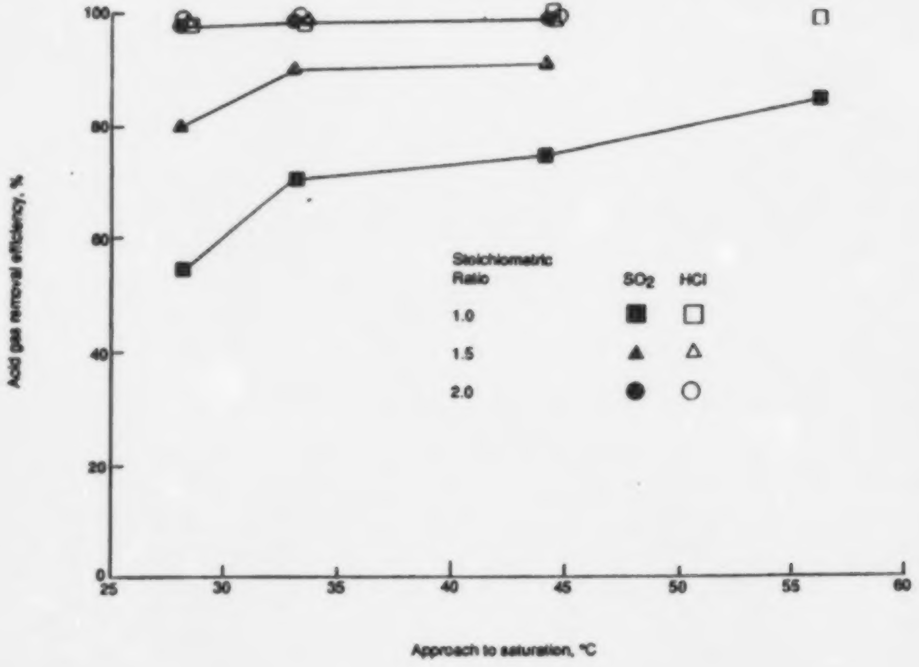

Figure 4. Acid gas absorption rates of sodium bicarbonate.

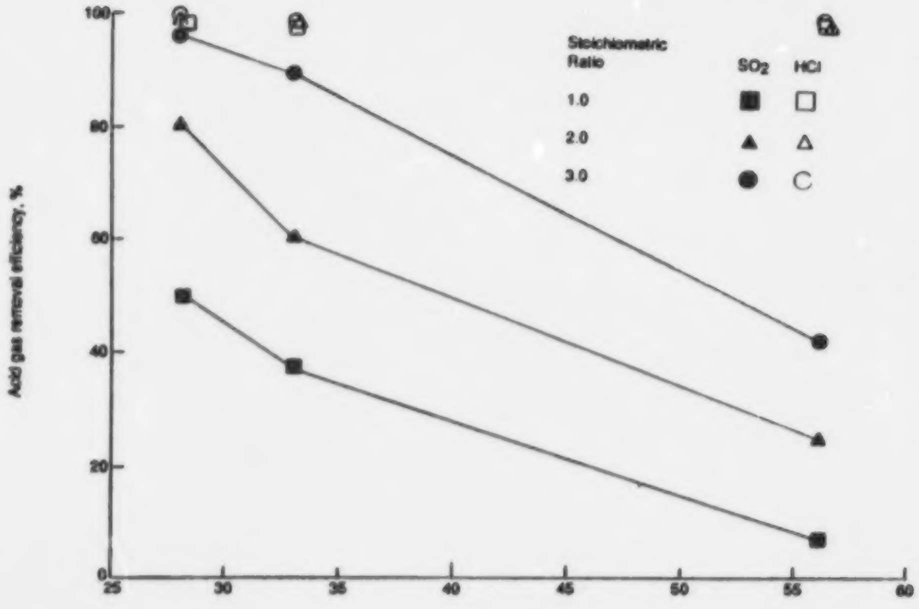

Figure 5. Acid gas absorption rates of magnesium hydroxide

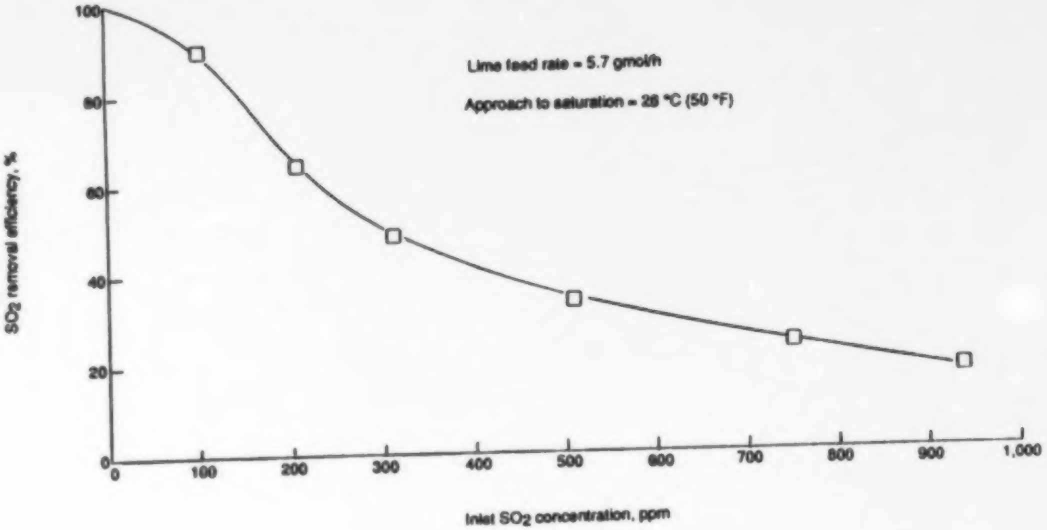

Figure 6. Baseline SO_2 removal efficiency of hydrated lime.

Lime feed rate = 5.7 gmol/h

Approach to saturation = 28 °C (50 °F)

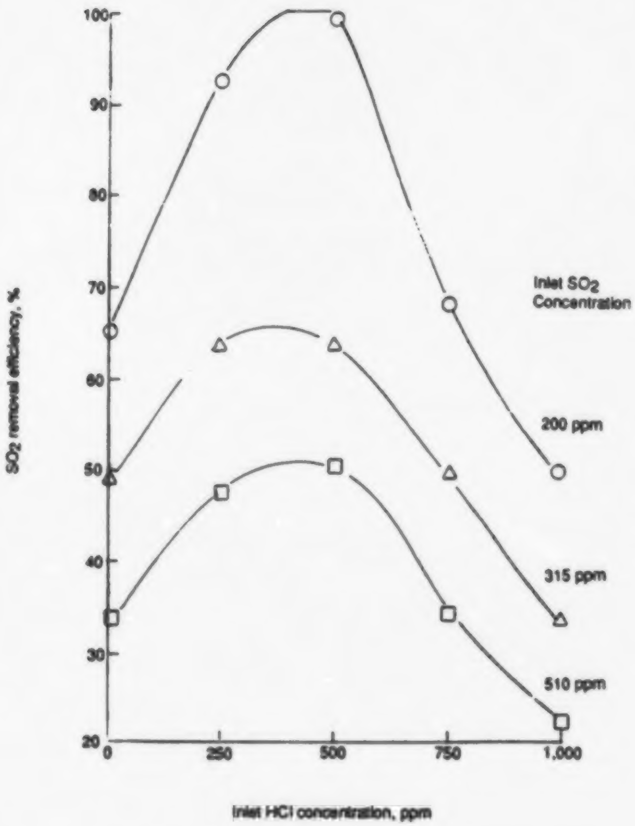

Figure 7. Effect of inlet HCl concentration on hydrated lime SO_2 removal efficiency.

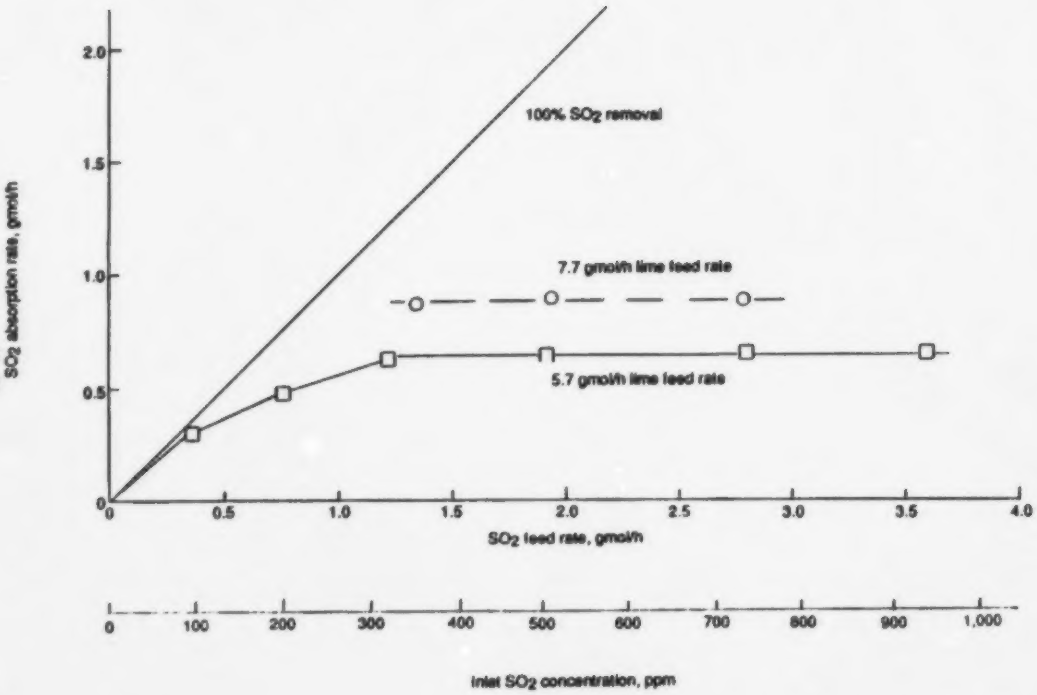

Figure 8. Baseline SO_2 absorption rate of hydrated lime at 28 °C (50 °F) approach to saturation.

MATERIAL RECOVERY WITH INCINERATION
MONMOUTH COUNTY, NEW JERSEY

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ABSTRACT

Monmouth County is located in east central New Jersey. The County landfill space will be exhausted by 1994 and the County is therefore planning for long-term disposal of solid waste. The County has developed an integrated program including recycling through source separation, household hazardous waste disposal, landfilling, and resource recovery. Based on public comment and a comprehensive technology assessment, the County selected the material recovery with incineration technology for the resource recovery portion of their solid waste disposal plan.

The material recovery process involves trommelling, magnetic separation and hand picking of targeted recyclable items. The waste material which is not recovered will be conveyed to the incinerator component of the resource recovery facility. Monmouth County has issued a request for qualifications for the facility and anticipates issuing a request for proposals in May, 1989. Construction start-up is currently projected for early 1991 with commercial operations beginning in early 1994.

INTRODUCTION

Monmouth County has a total land area of 471 square miles making it the sixth largest county in New Jersey (see Figure 1). All solid waste generated in the County is currently disposed of at a single site, the 291-acre Monmouth County Reclamation Center ("MCRC"), located near the geographic center of the County. The MCRC includes a landfill and shredding operation. At the current rate of disposal, the County anticipates that landfill space at the MCRC will be exhausted by 1994. The New Jersey Solid Waste Management Act requires each county in New Jersey to prepare a solid waste management plan for future disposal needs while upgrading existing disposal practices. In response to these requirements, and with strong political and public support, Monmouth County has embarked on an integrated approach to solid waste disposal including:

- Recycling through source separation and yard waste composting
- Household hazardous waste pickup
- Resource recovery
- Landfilling

In conjunction with the County and its resource recovery project manager, Birdsall Engineering, Inc., HDR conducted a review and assessment of 15 alternative technologies for solid waste disposal. In July, 1988, the County Freeholders selected the material recovery and incineration technology and provided authorization for initiation of the procurement of a facility with capacity of 1,700 tons per day ("tpd"). The objective of this paper is to describe the selected technology of material recovery and incineration.

CURRENT SOLID WASTE DISPOSAL PRACTICES

Since 1983, all processible and nonprocessible solid waste generated in the County has been disposed of at the MCRC, which is owned and operated by the County. Processible waste is waste that could be disposed of at a resource recovery facility including mixed municipal (household) waste, commercial waste and a small percentage of bulky waste. The County generated approximately 791,000 tons of waste in 1987, including nonprocessible and demolition waste. This figure decreased in 1988 to approximately 684,000 tons, partially due to the introduction of a mandatory source separation program in October, 1987. Processible waste generated in the County totalled approximately 487,000 tons in 1987, and 405,000 tons in 1988. The current tipping fee for waste disposal in the County (at the MCRC) is approximately $68 per ton.

At the 1987 rate of disposal, and with no changes in disposal practices, the County anticipates that landfill space will be exhausted by December, 1994. However, if a resource recovery facility can be developed and if recycling provides for a reduction of approximately 25 percent of the County's waste stream, this site may be sufficient for disposal of all of the County's waste for the next 25 years.

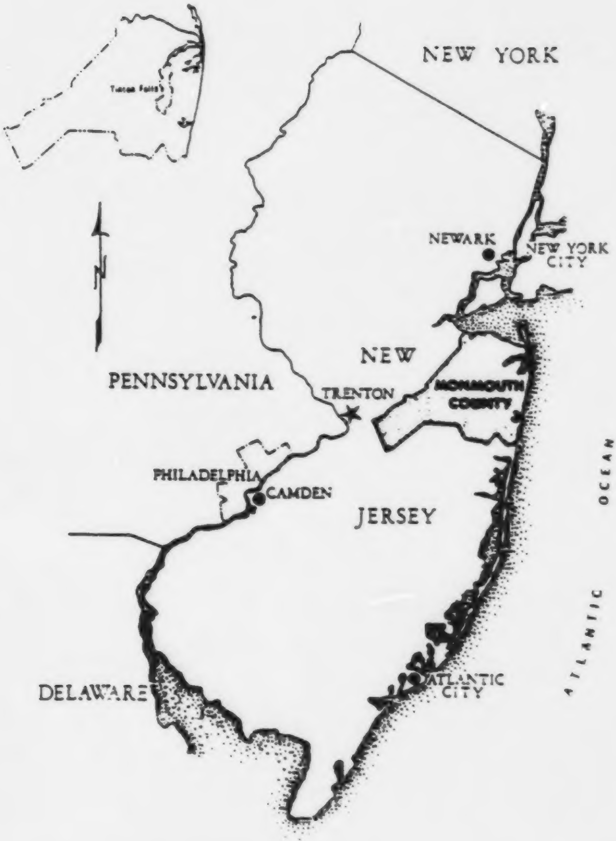
Tinton Falls
N
NEW YORK
NEWARK
NEW YORK CITY
NEW
PENNSYLVANIA
TRENTON
MONMOUTH COUNTY
PHILADELPHIA
CAMDEN
JERSEY
ATLANTIC OCEAN
DELAWARE
ATLANTIC CITY

Figure 1. Monmouth County, New Jersey

The County's solid waste management plan was amended in February, 1987 to include a mandatory recycling program. This program was the first to be certified under the New Jersey Mandatory Source Separation and Recycling Act of 1987.

The residential portion of the recycling program in Monmouth County was implemented on October 1, 1987. All residents are required to separate newspaper, glass bottles, aluminum containers, and tin and bimetal cans for recycling, and leaves for composting. Nonresidential establishments have also been required to recycle the above-mentioned materials, excluding leaves, plus corrugated cardboard and high grade office paper as of April 1, 1988. Recycling of asphalt, concrete, stumps and large tree parts were added to the recycling program as of October 1, 1988. The County also initiated efforts in 1988 to provide for the recovery and disposal of household hazardous waste through a program which includes drop-off locations and scheduled collection days. The County expects to achieve a greater than 25 percent reduction of waste through these recycling programs. Additional reductions of waste quantity will be achieved through the material recovery portion of the resource recovery facility.

WASTE COMPOSITION STUDY

The planning efforts for recycling and resource recovery included a study of the composition of the solid waste stream in the County conducted by HDR Engineering. The study consisted of five two-week field surveys over a one-year period, during which 335 samples of residential and commercial waste were obtained. Each sample was sorted into physical component categories which were selected using New Jersey Department of Environmental Protection guidance. The program was intended to provide more information on the amount of recyclable material in the waste stream, measure the success of the mandatory recycling program, and provide information for use in the technology selection.

The results of the field surveys show that quantities of some materials were significantly reduced in the residential waste stream during the course of the study, notably those that are recycled under the provisions of the mandatory source separation ordinance. Figure 2 graphically represents the quantities of some recyclable materials in the residential waste stream determined during the five field surveys. In the commercial waste stream, the only material designated by the County recycling plan showing a significant reduction in quantities landfilled was corrugated paper. Table 1 summarizes the weight percentages of each component in the combined residential and commercial waste stream. The composition shown in Table 1 is for the period following removal of recyclable materials by the mandatory recycling program and is anticipated to make up the waste feed for the resource recovery facility.

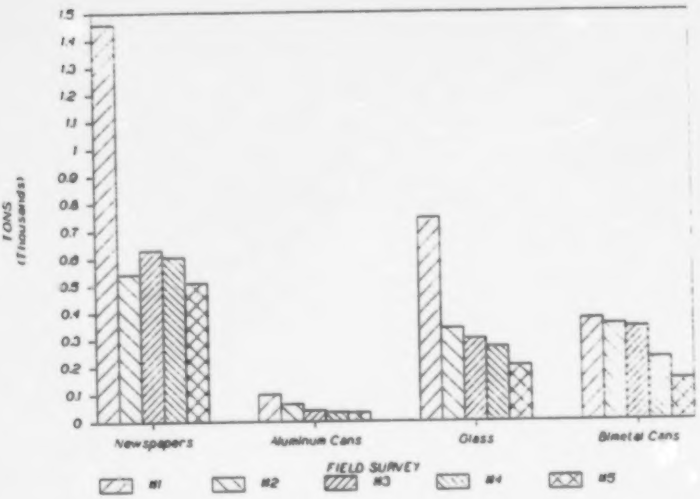

Figure 2. Residential Recyclable Waste Discarded

TABLE I. COMBINED RESIDENTIAL AND COMMERCIAL WASTE STREAMS

FIFTH FIELD SURVEY RESULTS[a]

Component	Weight Percent	Component	Weight Percent
Newspaper	5.7	Aluminum Cans	0.4
Corrugated	5.7	Other Aluminum	0.7
High Grade Paper	0.6	Nonrecyclable Ferrous Cans	0.5
Other Paper	34.1	Recyclable Ferrous Cans	1.3
PET	0.7	Other Ferrous	1.3
HDPE	3.0	Nonferrous Metals	0.4
Other Plastics	12.4	Brown Glass	0.3
Yard Waste	8.7	Green Glass	0.3
Food Waste	12.1	Clear Glass	1.6
Wood	2.2	Nonrecyclable Glass	0.2
Textiles & Other Combustibles	6.5	Ceramics & Fines	1.2
		Household Batteries	0.1

[a] Combined based on waste stream containing 2/3 residential waste, 1/3 commercial waste. Survey occurred July 11-23, 1988.

TECHNOLOGY SELECTION

The selection of a technology by the County involved a comprehensive review of available technologies and considerable public input. Vendor presentations were solicited through advertisements in national solid waste publications and 28 vendors responded. Public involvement in the selection process was gained through four public forums. Tours were conducted of 30 facilities in the United States and in Europe and were attended by project representatives. Finally, a review and assessment of 15 technologies were conducted, resulting in the recommendation of four technologies which were studied in detail.

The four technologies selected for detailed study by Monmouth County were:

- Material recovery with landfilling
- RDF preparation with grate-fired combustion
- Material recovery with incineration
- Mass burn

A detailed analysis of the costs for these facilities was conducted. In addition, the technologies were compared using the following criteria:

- Environmental impacts
- Landfill requirements
- Material and energy recovery capabilities
- Technology track record

The County selected material recovery with incineration as best meeting the established criteria. Many factors contributed to the County's decision. It is anticipated that the technology will provide for a reduction in the air emissions and an improvement in ash quality. The technology also provides the flexibility of targeting additional components in the waste stream which may, in the future, be determined to be either re-usable or possibly harmful. A significant reduction in landfill requirements will occur through recovery of materials, incineration, and reduction in incinerator residue. The material recovery with incineration technology is compatible with the existing mandatory recycling program. Finally, although this particular combination of material recovery and incineration has not been previously utilized in a facility of this type, it was felt that the material recovery system is simple enough in concept to avoid some of the problems encountered by more automated systems. The incineration component was considered a proven technology with high reliability. The County has initiated the procurement of a material recovery with incineration facility ("the Facility").

MATERIAL RECOVERY WITH INCINERATION PROCESS DESCRIPTION

In this application, the material recovery with incineration process will provide for the recovery of the following materials prior to combustion: aluminum; ferrous metals; corrugated paper (cardboard); film, HDPE and PET plastics; and household batteries. The system will utilize bag-breaking trommels, magnetic separation, and hand picking for material recovery. Waste that is not recovered will be sent to a combustion system. Steam produced from the combustion process will be used to generate electricity for in-house uses and for sale to the local utility. During the infrequent periods when the material recovery system is not operating, it will be possible to store the waste on the tipping floor or divert it directly to the incinerator storage pit.

The sizing of the Facility is based on the County's solid waste information, including information on the peak weekly and monthly deliveries to the MCRC. The Facility will have a rated annual processing capacity of 527,425 tons per year and a daily rated processing capacity of 1,700 tons per day. This rated capacity assumes the Facility will reach capacity in the year 2004, which is 10 years following Facility acceptance.

The waste receiving, handling, processing and combustion system is shown in Figure 3 and described in more detail below. Trucks entering the MCRC will be weighed at the County scale house. Trucks carrying processible waste (that which remains after the source separation program has removed recyclable materials amounting to an estimated 25 percent of the waste) will be weighed and will proceed to the tipping building. Trucks carrying nonprocessible waste will be directed to the landfill. Drivers will be directed by plant staff to the appropriate area on the tipping floor. The tipping floor will be separated from the incineration storage pit and tipping bays will be provided with closable doors for dust control. The tipping floor will be 200 by 350 feet in area, allowing up to 10 trucks to dump simultaneously.

Personnel will also be stationed on the tipping floor to identify and remove any nonprocessible or hazardous material. Corrugated paper will be removed from the waste on the tipping floor and fed onto a conveyor for baling. The bulky and nonprocessible wastes which are inadvertently accepted at the facility will be loaded by front-end loaders into 40-cubic-yard containers located in the auxiliary load-out area. These containers will be removed for the shredding and landfilling of the waste. The remaining waste will be pushed by front-end loaders into one of three processing lines. Each line will be designed to process approximately 45 tons of waste per hour and will be operated two shifts per day for 5-1/2 days per week.

A general schematic of the material recovery and processing system is presented in Figure 4. Waste material on the feed conveyors will pass by a corrugated paper hand picking station before entering one of three trommels. The rotating action and metal spikes in the first part of the trommel will be designed to break any glass, bags or boxes. The trommels will screen the waste into two streams. Five-inch holes will allow smaller materials such as glass, batteries, sand and grit, and most of the aluminum and ferrous cans present in the waste to pass to the "unders" stream. The ferrous will be removed magnetically and aluminum will then be hand picked from this smaller than five inches fraction. The ferrous stream will then be hand picked to remove household batteries. Remaining material that is not recovered after the hand picking and ferrous removal will be transferred to a feed conveyor which will discharge this material into the refuse storage pit.

The material larger than five inches will pass through the trommel ("overs") and be carried past hand pickers stationed along the conveyor system to collect PET, HDPE and film plastics. The facility will provide additional hand picking space for the removal of recyclable or other materials which may be identified in the future.

Each picker will be responsible for specific materials. Once taken off the conveyor belt, the material will be dropped through chutes onto conveyors that deliver the material to storage or to shredding or baling

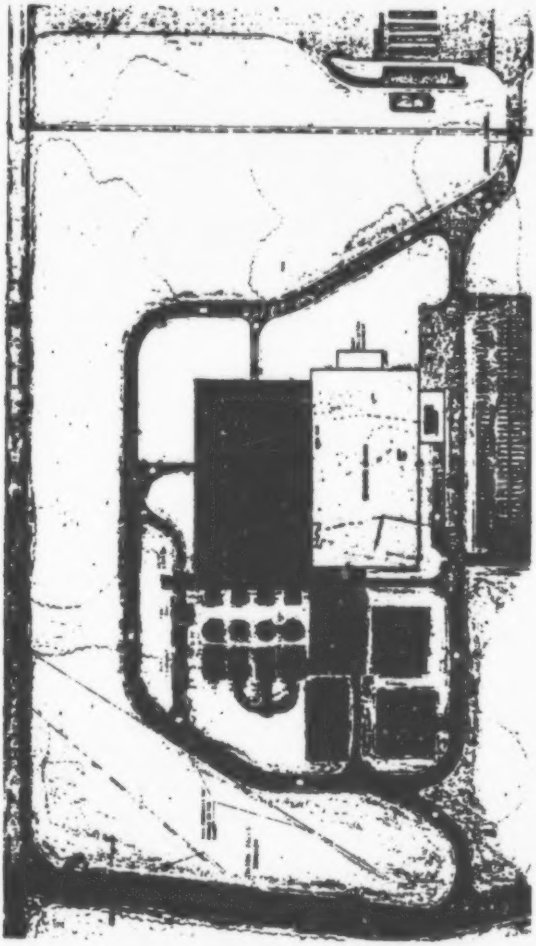

Figure 3. Facility Site Layout

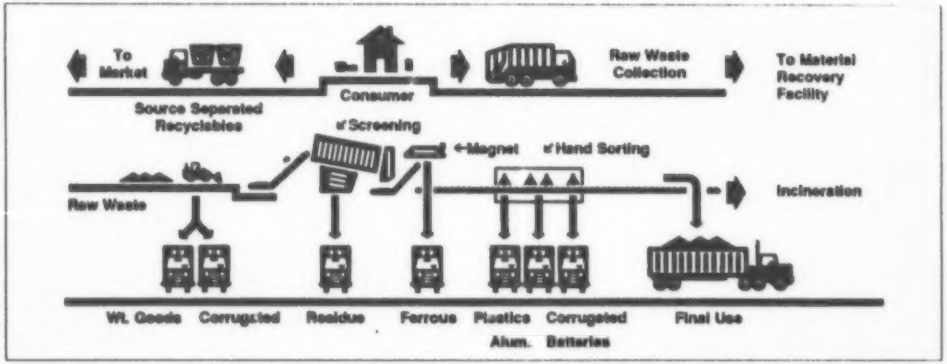
To Market
Source Separated Recyclables
Consumer
Raw Waste Collection
To Material Recovery Facility
Screening
Magnet
Hand Sorting
Raw Waste
Incineration
Wt. Goods
Corrugated
Residue
Ferrous
Plastics
Alum.
Corrugated
Batteries
Final Use

Figure 4. Schematic of Material Recovery and Processing System

equipment used for densification to facilitate handling and shipping. The waste remaining after hand picking of this "overs" stream will be subjected to additional magnetic separation prior to being transferred to a feed conveyor which will discharge this material into the refuse storage pit prior to incineration. The recovered ferrous is expected to have some contamination by materials which may need to be removed by hand picking or by shredding, depending on market requirements.

It is anticipated that the material recovery system will meet the recovery percentages for the targeted materials shown in Table 2. The percentages are on a percent-by-weight basis of the waste feed.

TABLE 2. RECOVERY PERCENTAGES BY WEIGHT OF INCOMING WASTE STREAM

Material	Approximate Recovery (Weight Percent)
Aluminum cans	65
Corrugated paper	45
Ferrous metals	80
Film plastic	40
PET plastic	40
HDPE plastic	40

This percentage removal would constitute an additional 5-10 percent reduction in the waste feed to the combustion system.

Waste discharged to the storage pit will be fed to one of three combustion lines by overhead cranes. Steam produced by the combustion process will be used to generate electricity for in-house use and sale to a utility. Flue gases will be cleaned using a scrubber and baghouse, according to state and federal regulations. Ash produced from the incineration process will be disposed of in a monofill at the MCRC.

MATERIAL RECOVERY SAFETY FEATURES

Material will be visually inspected prior to processing. As the waste is dumped onto the tipping floor, a waste spotter and the front-end loader operators will visually screen the waste for any material which could present a hazard to the pickers or equipment. Such wastes will be separated from the main waste pile and removed from the facility. The tipping floor and the processing plant will be separated from each other and enclosed, thus reducing dust levels in the processing building.

To protect pickers, picking stations will be located in separate enclosures which are well ventilated and air-conditioned. Workers will be required to wear protective gear, including hard hats, masks and heavy-duty gloves for pickers. Also, a training program will be developed to educate workers in the proper operation and safety features of the equipment.

Other safety and health features will include pull cords on each conveyor to allow for immediate stoppage of a conveyor at any time. If a conveyor in the processing line should fail to operate, all upstream equipment will automatically stop. Dust collection points will be located where waste transfers occur. Fire control equipment, including fire hoses and extinguishers, will be located throughout the plant. Plant personnel will be trained in all fire control procedures and the proper operation and safety features of all operating equipment.

CAPITAL COST ESTIMATES

The resource recovery facility capital costs will depend on the specific equipment chosen, and the bid price of the selected vendor. However, for the purpose of projection analysis, a construction cost estimate was prepared based on conceptual design criteria and is presented in Table 3. The actual construction price may vary by as much as 20 percent of this estimate.

The estimate is based on the following assumptions:

- 1,700-tpd facility capacity
- Three processing lines for the material recovery system and three processing lines for the combustion system
- Scrubber/baghouse for air pollution control
- Air-cooled condenser
- 40-MW turbine generator

TABLE 3. ESTIMATED CAPITAL COSTS FOR MATERIAL RECOVERY WITH INCINERATION TECHNOLOGY

Item	Amount
Capital Costs (1988$)	
Site Work	$ 11,000,000
Buildings	28,000,000
Processing Equipment	18,000,000
Combustion and Ancillary Equipment	112,000,000
Startup and Testing	9,000,000
Insurance and Other Nonequipment Items	14,000,000
Engineering (design, permit, and construction management)	17,000,000
Contingency (10% of capital costs)	21,000,000
Total	$230,000,000

The material recovery equipment and structures are estimated to cost $20,000,000.

OPERATION AND MAINTENANCE COSTS

The operation and maintenance costs for the material recovery with incineration technology were developed using the following assumptions:

- The processing facility will operate 88 hours during five and one-half days per week.
- The combustion facility will operate 24 hours per day, seven days per week.
- The operating capacity factor for the combustion plant is estimated to be 85 percent.
- 156 employees, including 75 hand pickers, will be required to operate the facility.
- Hand pickers will be paid $14 per hour.
- Aluminum; ferrous metals; corrugated cardboard; household batteries; and PET, HDPE, and film plastics will be recovered.
- All electricity (with the exception of electricity required to start the facility) required by the plant will be provided by in-house generation.

The total annual operating costs estimated for the facility, in 1988 dollars, are presented in Table 4.

TABLE 4. ESTIMATED OPERATING AND MAINTENANCE COSTS (1988$) FOR MATERIAL RECOVERY WITH INCINERATION TECHNOLOGY

Item	Cost
Operation and Maintenance (labor, admin., maintenance and materials, and misc.)	$13,600,000
Insurance	800,000
Utilities (water, sewer, and natural gas)	200,000
Residue Disposal	5,600,000
County Administration	300,000
Host Community Fee	500,000
Total Annual Operating Costs	$21,000,000

Of this total, approximately $10,000,000 per year (primarily related to labor) will be required for operation of the material recovery system.

REVENUES

Revenues obtained from the material recovery with incineration system will be derived from tipping fees, the sale of recoverable materials, and the sale of electricity. It should be understood that prices received for recovered materials may fluctuate dramatically and cannot be predicted on a long-term basis. The estimated prices per ton, in 1988 dollars, that could be obtained for recovered materials are shown in Table 5, along with anticipated quantities.

TABLE 5. ESTIMATED RECOVERED MATERIAL QUANTITIES AND REVENUES

Material	Market Value	Annual Tons Recovered	Annual Revenues
Aluminum	$1,000	1,600	$1,600,000
PET Plastic	100	2,500	250,000
Film Plastic	60	3,000	180,000
HDPE Plastic	100	1,600	160,000
Corrugated Paper	25	15,800	395,000
Ferrous	0	6,200	0
		47,350	$2,585,000

Electricity revenues from the facility are currently estimated to be approximately $11,000,000 during the first year of operation (1994). The projected tipping fee for the material recovery with incineration technology is estimated to be between $90 and $110 per ton (1994).

CONTRACTUAL CONSIDERATIONS

The County will require a single focus of responsibility and therefore a full service vendor, to design, construct and operate the material recovery and incineration facility. Some specific features of the full service contract will include the following:

- The vendor would be paid a separate annual operating fee or a per ton fee for operation of the material recovery system and the incineration system.

- The vendor will also receive 10 percent of the energy revenues from the incinerator portion of the facility.

- Staffing of the material recovery system hand picking operation will be the responsibility of the vendor. However, the salaries of the hand pickers will be treated as a pass-through cost. This is due to the limited labor supply in the Monmouth County area and the possibility that the County will request additional hand pickers to improve recovery efficiency.

- It is anticipated that the material recovery operations will be terminable on one year notice if the County feels the vendor is not using best efforts to provide for recovery of materials.

- Incentives to operate the material recovery system will be provided in the form of a percentage of material revenues.

IMPLEMENTATION SCHEDULE

Monmouth County authorized issuance of a request for qualifications for vendors interested in proposing on the material recovery and incineration system on December 8, 1988. Responses were received January 26, 1989 and the County is in the process of short-listing qualified vendors as this paper is being prepared.

The County has also identified its financial advisors and underwriters for the project and they are:

Financial Advisor:	Public Resources Advisory Group
Underwriters:	Shearson, Lehman, Hutton, Inc. First Boston Corporation Alex Brown Associates Jersey Capital Markets

The request for proposals for the project is scheduled to be issued in May 1989 with proposal responses due in August, 1989. It is anticipated that financing and construction will begin in early 1991 and facility operations will begin in 1994.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

REFUSE RAIL HAUL: THE MISSING LINK TO RESOURCE RECOVERY IN MONTGOMERY COUNTY

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ABSTRACT

A novel type of intermodal container train concept has been developed for Montgomery County, MD. Its purpose is to haul refuse from a transfer station to a remotely located processing plant for electrical power generation. Residue needs to be back hauled for disposal in a landfill at a third location. In the search for environmentally acceptable technology, operational European rail haul systems are evaluated. During a conceptual design study, it was determined that the optimum combination would result from integration of European container technology with American rail car technology. Pertinent technical parameters are compared and reference is made to present project procurement activities.

BACKGROUND AND INTRODUCTION

During the development of resource recovery projects, proper facilities siting becomes a critical issue. This is especially true in the case of refuse power plants (RPP's) where the preferred energy customer and the waste collection centers are in different geographic locations. Normally, transfer hauling by truck is the means to overcoming these obstacles. However, in some localities, the existing roads are inadequate and the citizenry objects to the added noise and pollution associated with trucking.

This situation is further exacerbated by political realities which mandate a two-way haul system, where refuse travels in one direction and ash in the other. Further complications arise from the need to be able to store refuse and/or ash in an environmentally safe manner in multiple locations in order to smoothen the flow of material.

Montgomery County in Maryland has encountered all of these problems during planning of its RPP sized for an initial capacity of 1,800 STPD, and 53 MW. These problems were overcome by opting for a two-way rail haul system. Before making this decision, studies were conducted for the purpose of learning from the experience with commercial refuse rail haul elsewhere.

From these studies, the Schwandorf project in Bavaria, West Germany, emerged as an example of successful refuse rail haul on nearly the same scale. Consequently, a similar technical approach was targeted for Montgomery County. However, it was then determined that new American railroad equipment, i.e. the double stack cars, had entered into service. This would permit an even higher density of refuse rail haul than the one practiced in Europe. As a result, European and American refuse rail haul technologies were merged for an integrated system, which has now entered into the procurement process.

PROJECT OBJECTIVE AND CAPACITY DETERMINATION

On July 7, 1987, the Montgomery County Council adopted Resolution 11-382, which states in part:

> (1) The central disposal facility for the County's solid waste will be a mass-burn resource recovery facility located adjacent to the Pepco generating station near Dickerson.
>
> (2) The method of transport of solid waste from the existing transfer station to the new facility, and residue from the facility back to the transfer station, is by rail.

On the same date, the County joined the Northeast Maryland Waste Disposal Authority as a participating jurisdiction. Thus the Authority and its team of consultants were authorized to begin project implementation.

Figure 1 illustrates the various parts of the transportation system in the form of a flow schematic. As planning got underway, one of the first problems encountered was sizing of the project.

Although the County produced a dependable historical data base on waste collection, it became rather difficult to predict the future. To some extent this was due to uncertainty over just how successful the County's recycling program was going to be. Furthermore, the next census is another year away and current population statistics are hard to find.

In order to determine the capacity of the transportation system, a statistical analysis was performed. Working with a 5-year data base, averages were calculated for weekly tonnages, one for each week of a normalized year. Certain adjustments were made in order to deal with non-processible waste components, to compensate for abnormalties and to consider the effects of recycling. See Ref. (1).

Figure 2 shows the results of these derivations in the form of a cumulative frequency diagram. Both the actual and the normal distributions are graphed. The upper control limit at the 96% confidence level was also calculated. This limit was then rounded up to 12,000 STPW (short tons per week) which was then set as the design point for the refuse part of the transportation system.

Since it was decided early in the project that ferrous scrap should be recovered at the facility, the net residue rate for the return haul was to be 30%. This was then equated with 3,600 STPW as the design point for the residue part.

DESIGN AND OPERATING CRITERIA

It would go beyond the scope of this paper to discuss the many requirements which were applied to development of the Montgomery County Refuse Rail Haul System. However, several of the more salient design and operating criteria are listed in the following:

1a. Design Tonnage for MSW: 2,000 STPD$_6$, 12,000 STPW$_{52}$
624,000 STPY

1b. Design Tonnage for Residue: 600 STPD$_6$, 3,600 STPW$_{52}$
187,200 STPY

2. Single unit train, rotated daily, will accommodate entire design tonnage.

3. Unit train must not interfere with commuter traffic on main track, i.e. operations are only permitted during certain "windows."

4. Single CSX crew must be able to operate unit train within basic working day of 8 hours plus 2 hours of overtime.

5. CSX crew must be scheduled for day shift. Minimization of overtime is highly desirable.

6. Pull weight of unit train must be minimized in order to limit motive power to a single locomotive. Train sections must be compatible with robot locomotives during indexing operations.

7. Pull length of train must be minimized in order to allow manuevering in small stations. Land use requirements must be reduced to the smallest possible amount.

8. Number and lengths of tracks, number of switches and signals must be held to a minimum in both stations and must be compatible with site constraints.

9. Vehicular ingress and egress at transfer station must be limited to daytime hours. (8:30 AM to 4:15 PM Monday through Saturday)

10. Complete unit train must be loaded and unloaded at transfer station within a single 3 1/2 continuous time period. Complete turn-around of unit train must be finished in 4 hours or less.

11. All refuse must be removed from the receiving building at the transfer station before midnight each day.

12. Containers must be totally enclosed in order to prevent leakage and/or spillage. Containers must be environmentally acceptable and safe for operating personnel.

13. Containers must be structurally sound to permit stacking of three high and to tolerate the impacts associated with crane and rail movement in both the full and empty condition. Containers must resist damage from accidents.

14. Containers must be a two-way transport device for MSW in one direction and residue in the oppposite direction.

15. Containers must be of the intermodal type which can be moved legally over rail and road. Must be compatible with fully mechanized hoisting equipment.

16. Manpower needs must be minimized by resorting to a high degree of mechanization and automation. Conversely, personnel productivity is to be maximized.

17. Containers and ancillary equipment must be fully weatherized in order to permit operation under all conditions of inclement weather.

18. Major equipment must be maintainable at least for 20-year service life. Equipment replacement schedules must be minimized.

19. Technology must have been in full-scale commercial operation for a continuous 1 year period, or more.

20. Performance of technology and logistics plan must be guaranteed.

21. Equipment and operations must be acceptable to rail carrier, i.e. CSX Transportation Company.

22. Certain structures already existing at the transfer station site (such as the waste reception building and the citizens' drop-off facilities) must be retained.

PREVIOUS ATTEMPT AT RAIL HAULING IN MONTGOMERY COUNTY

The idea of building a central waste processing facility or CWPF in Montgomery County is not a new one. In 1975, the County adopted its first "Ten-Year Solid Waste Management Plan" which among other measures called for construction of a CWPF. The selected site is a 41-acre piece of property located at Shady Grove Road near Rockville, Maryland. See Ref. (2).

The plan called for the County to construct, own and operate the CWPF for the purpose of recovering materials such as scrap metals and glass. However, the main product was to be a refuse-derived fuel or RDF. In fact, a Memorandum of Understanding had been agreed upon by the County and the Potomac Electric and Power Company (Pepco).

Under the provisions of the memorandum, Pepco was to donate to the County 17 acres at the site of its power generation complex at Dickerson for building a RDF receiving and unloading facility. In addition, Pepco would permit construction of a rail line across its property to service sidings to be constructed as part of the RDF unloading facility. Pepco also intended to furnish a conveying system which would transport the RDF to its boiler room. Pepco planned to accept "...any quantity of RDF up to 10% of the average heat input into its boilers as defined in BTU's..." Yet, Pepco retained in its "sole judgement" the right to refuse acceptance of RDF delivery at any time if there were adverse effects on its equipment performance. This included both its boilers and air pollution control equipment. See Ref. (3).

Market studies undertaken at that time led the County to decide to eliminate glass and non-ferrous metals recovery because of adverse economics.

Figure 3 shows a layout of the proposed design. The design criteria called for a 1985 MSW processing rate of 1,686 STPD with an RDF production rate of 1,335 STPD . A 3-shift operation was recommended, with 2 shifts for operation and 1 shift for maintenance.

Steel piggy-back containers of 75 yrd^3 volume capacity were selected as the basic transportation unit. Since the RDF was expected to be of the fluff type, stationary compactors were to increase bulk density from 4-6 to 22-30 Lb/ft^3 when charging RDF at a rate of 24-40 stph. Accordingly the minimum commodity load per container was to reach 23 ST. A total of 259 containers and 115 railroad flat cars were to be purchased.

Based on a 10'x10'x20' box shaped configuration, 2 containers could be placed on each flat car, leading to a daily train of about 30 cars and 60 containers. A quantity of cars and containers were to be held in reserve.

The Baltimore & Ohio Railroad (B&O, now CSX) already provided freight service to the Sears & Roebuck warehouse which straddles the northwest boundary of the property. Therefore, it would be convenient to build a rail yard which would be connected by switches to the existing Sears siding. During discussions with the County, B&O indicated that it would take about 3 days on the average to turn a single train around.

In order to satisfy the volume and weight requirements discussed above, a relatively large rail yard was required. A total of 15 tracks were arranged in 3 sets of 5 parallel tracks each. Tractors with flat bed trailers were to haul the containers from the CWPF to the rail yard. Here, a rubber-tired straddle crane was to pick the containers off the trailers and put them on the flat cars. Empty return containers were to be handled in a similar fashion.

This arrangement required a lot of space, which is evidenced by the fact that 54% of the 41-acre total was to be occupied by the rail yard, buildings and parking lots.

In the final analysis, the County decided not to implement the proposed CWPF because of the high risk associated with RDF technology at the time. In addition, Pepco opted not to construct a new fourth boiler, especially equipped for firing RDF.

SEARCH FOR NEW TECHNOLOGY

Thus far, rail haul systems for MSW which meet Montgomery criteria, are not in commercial-scale operation in the U.S. On the other hand, several large scale systems have been operating successfully in Europe for a number years.

Rather than wanting to "reinvent the wheel," we went to Europe in order to inspect the most representative systems. See Ref. (4). In the text below, some of our impressions are summarized.

(1) Schwandorf

Located in Eastern Bavaria, FRG, this is a regional rail haul system for transporting raw MSW from 8 separate transfer stations to a single MSW refuse power plant (RPP) built on the property of a large aluminum production plant. This RPP has installed capacities for refuse processing and power production of 3x20.63x24=1,485 $STPD_7$ and 2x11.6=23.2 MW, respectively.

Operating commercially, approximately 1,370 $STPD_5$ of raw MSW are brought in 5 days per week. Additionally, 127 STPD are brought in from the host community by direct truck haul. The transport media are high density 8'ø x 30'L ISO containers which are placed on railroad flat cars in sets of 2 each.

Two dedicated trains as well as one shared train are used to cover the disposal region on a daily basis. (Note: One train moves along the N-S axis while the other follows the W-E axis with Schwandorf being located at the cross point. Both trains are cycled during a given 24-hour interval. Thus each set of transfer stations is served by a single train. Since the aluminum smelter receives considerable shipments of other bulk cargos, the refuse container trains must not interfere with other operations.

The unloading operation at Schwandorf is automated to a fair degree, and it includes the deployment of a robot locomotive which indexes the rail cars at the work station. This locomotive is radio-controlled by the crane operator who also unloads and reloads containers. He rides in the crane's cab. A second operator is positioned on the elevated unloading ramp where containers are moved by sleds to the refuse bunker wall for hydraulic discharging.

Only 28 people are needed to handle the containers and the waste at the 8 transfer stations and at the RPP itself. At all 9 locations, certain peripheral duties such as scale-house operation and groundskeeping are included. This equals an overall personnel productivity of 12,700 ST/Person-Year. This system must operate outdoors in all weather conditions on a year-round basis. All of the original containers are still in service today.

About 152 containers and 38 rail cars are provided, including some spares. Thus, there are 2 complete sets of containers and only 1 complete set of rail cars. Accordingly, the ladder truck system has a minimum of three tracks operational: two for loading and/or unloading containers and one for running the engine around.

The containers can be stacked up to three high, empty or full, depending on logistical requirements at the transfer stations. The charging presses can process a variety of wastes of residential, commercial and industrial origin. Bulky wastes are also easily accommodated. The same containers are compatible with truck hauling which is practiced elsewhere, i.e. they are truly intermodal.

The compactness, efficiency and size of the whole operation is impressive. Plans are now underway to expand the operation so that an additional 600 $STPD_5$ can be processed.

Although containers are weighed during the charging process in order to optimize payloads, there are some variations from one container to the next. We are are still investigating the range of this variability as well as the underlying causes.

Test data from three different transfer stations shows that the average commodity load per container was 25.95 ST with control limits of 21.45 to 30.45 ST at the 95% confidence level. The coefficient of variation was only 8.7%, which also indicates a close measure of control.

Collaboration with the federal railway system has been excellent from the beginning on, and Schwandorf must be judged a success by any standard. Stationary hydraulics are used both at the RPP and the transfer stations in order to maximize payloads.

(2) Bamberg

Presently, installed capacity at this codisposal refuse power plant (C-RPP) is 475 $STPD_7$. During 1986, 170,800 ST of MSW were actually incinerated. In addition, 7,714 WSTPY @ 35% TS of sludge cake were incinerated as well. Of the total annual MSW tonnage delivered, about 61.6% was delivered by truck and 38.4% by rail/truck transfer. The latter method means that after unloading from the rail cars, the containers are placed on a trailer chassis for a short ride to the C-RPP.

While the C-RPP has been in operation since 1978, the rail haul system was added later in 1980 so that the available plant capacity could be better utilized by bringing in waste from outside the host community. The containers are 8'Øx30'L, in accordance with ISO requirements. On the average, 23.10 ST/unit of commodity load are achieved.

Unloading from the flat car, transfer by truck and discharge into the refuse storage pit is accomplished by 1 person working about 3-4 hours each day, 5 days per week for a total of 20 hrs/week. He handles 6 rail cars and 12 containers on a typical day. During 1986, total receipts by rail transfer totaled 65,600 STPY. This translates into a personnel productivity of 118,600 ST/Person-Year for the unloading operation. At the transfer station an additional 5 persons are employed for a personnel productivity of 13,120 ST/Person-Year. Thus, overall personnel productivity for all operations, i.e. loading and unloading runs presently at 11,860 ST/Person-Year.

A simple ladder track system adjacent to the Main River is used for positioning of the container train. For the container loading and unloading operation a single, dead-ended side spur is used. At the transfer stations, stationary hydraulics are used, while unloading at the C-RPP is done with mobile hydraulics. Initially 36 containers and 6 rail cars were purchased. A single train section is used to deliver full containers and return the empty ones during the same 24-hour period. Some minor indexing can be done by the gantry crane itself, which is tracked. However, all major indexing is done by means of an electrically driven winch.

The German Federal Railway System has been a very reliable partner and its delivery service has never failed. Container storage for 3 to 4 days is common, although once a 2-1/2 week storage period occurred without odor problems.

Boiler operators in the C-RPP prefer the wastes which are discharged from the containers because of some homogenization and size reduction which take place during charging and discharging. Thus in the case of mass burning, combustion control may be improved. No serious operating problems have been reported, although the operator would prefer some protection against the weather.

(3) Kempten

Designed by Altvater, this oversized rail container system links the Sonthofen transfer station with the Kempten RPP in Southwestern Bavaria. The one-way distance is about 20 miles and 33,000 STPY of MSW are to be transported. This system is one-of-a-kind, and has not been repeated anywhere else in the world. Altvater had bid unsuccessfully on the Schwandorf and Bamberg projects.

Originally advertised as capable of loading payloads of up to 45 ST per unit, this was hardly ever accomplished in routine operations. (In practice, only 34 ST per unit was averaged with the original containers.) Ultimately, the containers had to be redesigned to increase their volume from $73m^3$ to $100m^3$ in order to make the guaranteed unit payloads. These containers are the only ones which have a real pressure seal, i.e. a giant O-ring, which permits a vacuum-tight closure. Thus, completely odor-free storage for periods of 4 weeks, either at the work stations or in transit, is possible.

These containers do not comply with ISO standards and they are not truly intermodal, either. For demonstration purposes, they have been hauled over land by means of special heavy-duty tractor-trailer combinations. Special truss-type rail cars are required as well. These containers are heavy and awkward to move. Rather than cranes, a system of cables and pulleys is used. During loading and unloading, they must be rotated slowly in order either to pull waste to the inside or to eject waste to the outside. In spite of this apparent complexity, this system is fully mechanized and requires very little labor. Essentially 1 man performs all functions, yielding a personnel productivity of 106,000 ST/Person-Year for the unloading operation. Overall personnel productivity is estimated at 13,200 ST/Person-Year.

Other than the original failure to achieve maximum payload, no major operating problems have been reported.

(4) Troisdorf

Designed for a maximum daily capacity (Monday - Friday) of 992 $STPD_5$ (or 165 stph), this is the latest of the rail haul systems to enter into service in Germany. The purpose is to collect waste from the Rhein-Sieg County and its municipalities and to transport it by rail to a distant landfill (about 38.6 miles away one-way). This is the near-term objective, but for the long-term, an RPP will be constructed somewhere at a distant location. Actual daily throughput is projected to average out at 775 $STPD_5$.

Although square in cross section, these containers fit ISO dimensions and are suitable for truck transport as well. As at Schwandorf, a robot locomotive is used for train indexing. Only a single crane track with one spare motor is provided (Note: Schwandorf has a spare crane on the same track). The operational reliability of this arrangement may be questioned by some, but judging by the Bamberg and Schwandorf experience, no major time-consuming breakdowns should be encountered.

The crane arrangement is such that even with containers stacked three high, there is enough height to allow for the clear passage of a 4th container above.

A crew of 8 operates this transfer station on a daily basis. Besides the crane system, 6 charging presses (with 2 control rooms) and 2 truck scales need to be operated by this crew. There are also certain housekeeping chores which need to be carried out. Because of a substantial presence of bulky waste, more presses than normal were provided. However, the base crew for the compaction and loading activities consists of only 3 persons. In addition, there are the positions of yardman, floater and plant manager. Thus a personnel productivity of 25,200 ST/Person-Year is planned for the loading operation. At the final destination, another 3 people are projected for unloading and transfer to the landfill. Thus, overall personnel productivity is estimated to amount to about 16,800 ST/Person-Year.

Commercial operation commenced on December 1, 1987 and thus far no serious problems have been reported.

Charging of containers can be accomplished by either one of two methods: (1) direct charging in the case of residential waste, and (2) prebaling in the case of certain commercial or bulky waste. Container loads were tested to average 21.03 ST although up to 24.24 ST were guaranteed in the performance specifications.

Because of their rectangular cross section, these containers have a better space utilization and require less compaction of their payloads. This is thought to be especially beneficial for bulky wastes.

The ladder track system leading to and from the transfer system is curved. Therefore, a diesel-powered robot locomotive rather than an electrically-powered unit is used for indexing. (Note: Electrical robots are usually connected by a trailing cable.) A typical unit train consists of 18 flat cars with 36 containers. Interestingly enough, a single train is used during the course of the day and not two. This is possible because of the speed of the operation and interim storage at the transfer station. A total of 85 containers and 20 rail cars were purchased.

The ladder track system has only two parallel spurs, which means that the Federal Railroad must break up each train into two sections. On arrival, these sections must be pushed rather than pulled because there is no run-around track for the locomotive to get past the cars.

(5) Winterthur/Zug

Located in northern Switzerland, this RPP has an installed capacity of about 700 $STPD_7$. On the average, approximately 108.6 STPD is received via rail haul, while the majority comes by direct haul. Based on actual 1986 statistics, the RPP incinerated a total of 100,282 ST of refuse. Out of this total, 28,232 ST or 28.2% were delivered by rail.

This is a rather simple and flexible system. It is totally different from the systems previously described. However, it does not meet Montgomery criteria. Therefore, it was eliminated from further consideration.

Parallel to our study, the Southern California Association of Governments performed a similar study. See Ref. (5). In particular, several large refuse rail haul systems in England and Holland were investigated. Although these systems had multi-year operating experiences, we found that none of them could satisfy the Montgomery criteria. Therefore, they are not discussed any further within the context of this paper.

In terms of operational efficiency, environmental compatibility and personnel productivity, we concluded that the systems operating in Bamberg, Schwandorf and Troisdorf, West Germany, were promising candidates for the Montgomery County application. But were there significant differences?

In order to answer this question, we performed comparative analyses, the results of which are contained in Tables 1, 2 and 3. Inspection of these tables suggests that commodity-to-payload ratios of 0.75 to 0.81 should be attainable together with commodity-to-rail car weight ratios of 0.58 to 0.61. In addition, rolling resistances of approximately 4.1 Lb/ST should be attainable as design goals. Likewise, personnel productivities in the range of 12,000 to 18,000 STPY/Person should be realistic.

CONCEPTUAL DESIGN STUDY

After the technological reviews, the next logical step of project development called for a conceptual design study. This study was to serve three purposes simultaneously: first, to define major transportation parameters on which a technical and economic feasibility study could be based; second, to serve as a focal point for the Authority's project team which was to prepare procurement documents; and third to furnish a basis of negotiations for the transportation agreement.

In the first round of this study, candidate European and American container concepts were mated with rail cars available from domestic sources. The immediate objective was to identify systems which would offer the best combination of minimum pull length and pull weight for same number of ton-miles per year.

The results are delineated in Table 4. For comparison purposes, both European and American rail equipments were included. In the final analysis, the "American Super Train" emerged, which involves high density 8'øx30'L containers of European design integrated with articulated 5-pack double stacking rail cars of American design. With a CTCWR = 0.63 it is slightly better than Schwandorf, but with regard to SCL of 1.16 it is a great deal better than Schwandorf. In fact the "American Super Train" is expected to be only about half as long. In addition, the "American Super Train" is projected to have a higher personnel productivity with 18,350 STPY/Person employed. For illustration of a typical U.S. container train with articulated double stack cars, refer to Figure 4.

This concept can be viewed as revolutionary once its impact on the rest of the transportation system is fully understood. For brevity's sake, only a few advantages are mentioned here:

-Space for movement and storage of equipment is reduced.

-Track construction requirements are decreased.

-Indexing operations are shortened.

-Train turn-around time is diminished.

It must also be pointed out that both key pieces of equipment, i.e. the high density containers and the double stack cars have already been proven in multi-year service on an individual basis. What is new about the "American Super Train" is the combination of the two.

Two back-up concepts were also developed. Of these, the more promising one is based on modified 8'Wx8'Hx40'L box containers placed in double stack articulated cars.

During the second round of the study, different logistics plans were developed. These were passed onto CSX for review and comments. In addition, CSX furnished its own requirements for design and operation of railroad equipment. One of the sticky points concerned approval of 125 ton truck assemblies by CSX Engineering. These extra high capacity assemblies are somewhat newer and may lead to increased track maintenance. Also, there were questions concerning permissible track clearances, but in the end they were all resolved.

FEASIBILITY STUDY

On a parallel path to the conceptual design study, an economic feasibility study was performed. See Ref. (6). Since this study has not been released, it is not possible to furnish a detailed discussion.

Basically three generic technologies were reviewed in terms of their design, operating, and cost characteristics. According to the volumetric reduction achieved during container loading, they were classified:

low density (LD) system	1:1 to 2:1
medium density (MD) system	2:1 to 4:1
high density (HD) system	4:1 to 6:1

Capital and operating costs were compared on a 20-year life cycle basis. Generally, conservative assumptions were made in most respects. The costs of capital and depreciation were included. On the other hand, the cost of land was disregarded.

The conclusions suggested the feasibility of both the HD and MD systems, with a slight edge for the HD system. Since detailed designs were not furnished, it was not possible to fully assess the site impact of one system over the other. However, in view of the discussions in the preceding sections, the HD system was considered to be the most compatible with existing site constraints.

TRANSPORTATION AGREEMENT

A draft agreement has been completed which is acceptable to both the Authority and CSX Transportation. See Ref. (7). This is a prototype agreement for 20 years of dependable rail service, moving MSW in one direction and residue in the other. It involves performance guarantees and penalties for both parties. We believe that it may be the first of its kind in the nation. However, since this document relates to the procurement which is in progress, it has not been released by the Authority as yet.

PROCUREMENT PROCESS

On February 15th, 1989, the Authority formally released a Request for Proposals (RFP) to interested parties. See Ref. (8). This 3-Volume document states the technical, financial and environmental requirements for both a resource recovery facility and a refuse transportation system. A private entity is sought to design, build and operate the project, including the rail transportation system. In a departure from previous practice, the Authority will own the project.

In one appendix, the technical requirements for the transportation system, including the rail haul part, are discussed. By and large, these are the minimum requirements and they use the results of the concept study as the framework. However, the respondents are free to propose alternate concepts as long as they can meet certain performance and qualification criteria.

Since this procurement is underway, it is not appropriate for us to comment any further.

CONCLUSION

1. From the studies undertaken over a 2-year period, a design concept for refuse rail haul has emerged which promises to set new standards. By integrating technologies advanced in Europe and the U.S. a high degree of cleanliness, efficiency, precision and reliability of refuse container service by rail should be attainable.

2. Of course, during the procurement process, this concept will be challenged by others. In order to survive the competitive process, it must demonstrate its cost effectiveness. However, from the viewpoint of operational density, it will be a concept which may be hard to beat.

3. Compared to the early attempt in 1975, the new rail haul concept shows dramatic improvements. For example, the commodity load was increased by 50% while train turn-around time was decreased from 72 to 8 hours. Instead of 15 tracks, only 3 tracks will be required at the transfer station. Instead of 115 flat cars only the equivalent of 45 double stack cars will need to be bought.

4. Perhaps the single most important aspect of hauling refuse by rail is that no new heavy truck traffic will be added to already congested highways. In fact, some of the existing heavy vehicular traffic between the transfer station and the landfill will be eliminated.

5. Rail hauling of refuse and residue has allowed the Montgomery County project to go forward after long periods of stagnation. Thus, it has become the missing link. However we do not expect a proliferation of refuse rail haul systems in North America in the near future.

6. It is hoped that the pioneering efforts of the Montgomery County project will open up a new avenue to other resource recovery planners who appear to be bogged down with seemingly insurmountable siting and transportation problems.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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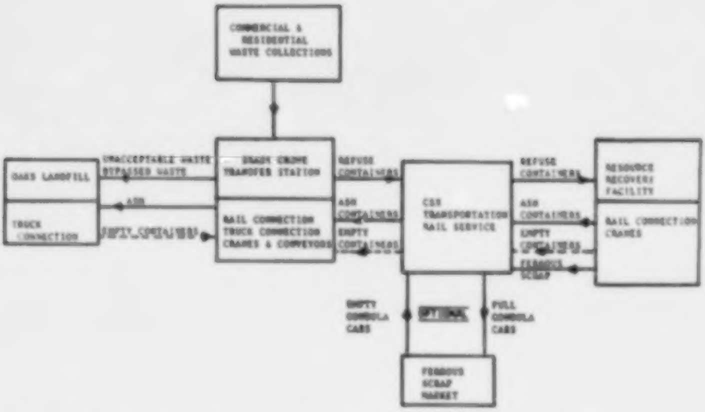
COMMERCIAL &
RESIDENTIAL
WASTE COLLECTIONS
OAKS LANDFILL
TRUCK CONNECTION
UNACCEPTABLE WASTE
BYPASSED WASTE
ASH
EMPTY CONTAINERS
SHADY GROVE TRANSFER STATION
RAIL CONNECTION
TRUCK CONNECTION
CRANES & CONVEYORS
REFUSE CONTAINERS
ASH CONTAINERS
EMPTY CONTAINERS
CSX TRANSPORTATION RAIL SERVICE
REFUSE CONTAINERS
ASH CONTAINERS
EMPTY CONTAINERS
FERROUS SCRAP
RESOURCE RECOVERY FACILITY
RAIL CONNECTION CRANES
EMPTY GONDOLA CARS
OPTIONAL
FULL GONDOLA CARS
FERROUS SCRAP MARKET

Figure 1. GENERAL SCHEMATIC OF REFUSE TRANSPORTATION SYSTEM

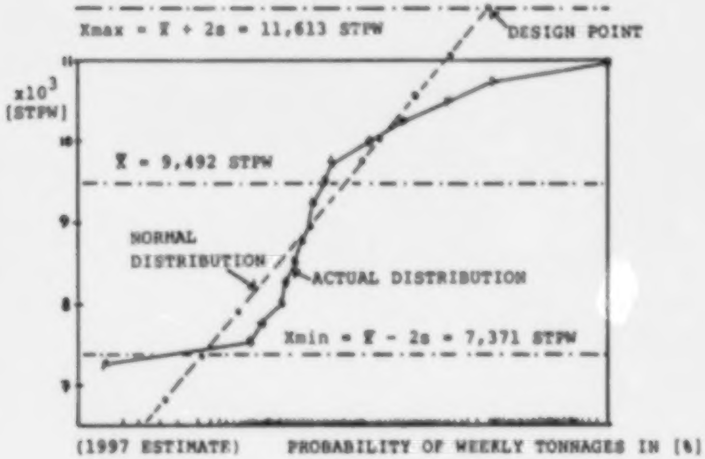

Figure 2. CUMULATIVE FREQUENCY FOR WEEKLY TONNAGES OF PROCESSIBLE WASTE IN MONTGOMERY COUNTY (1997 PROJECTIONS)

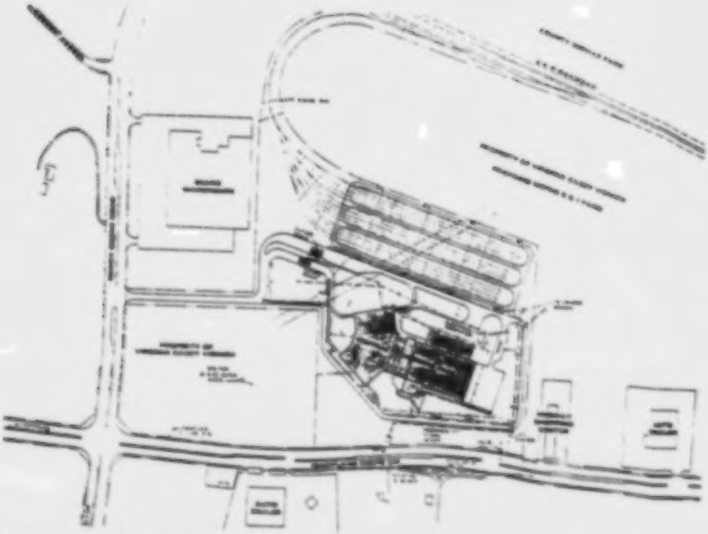

Figure 3. PROPOSED CENTRAL WASTE PROCESSING FACILITY WITH RAIL HAUL SYSTEM MONTGOMERY COUNTY, 1975

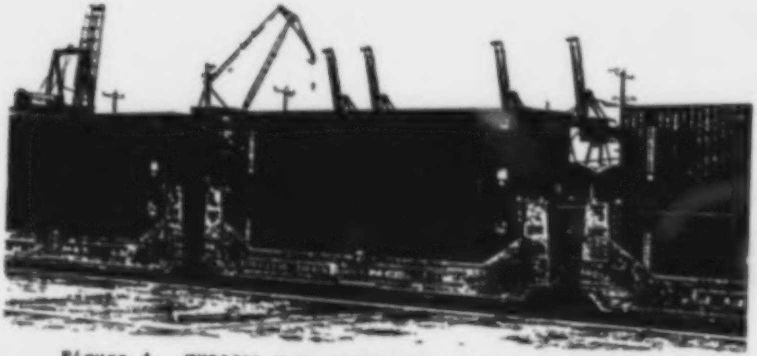

Figure 4. TYPICAL U.S. CONTAINER TRAIN WITH ARTICULATED DOUBLE STACK CARS

Table 1. COMPARISON OF DESIGN PARAMETERS FOR MSW RAIL HAUL CONTAINER SYSTEMS OPERATING IN EUROPE - Container Technology

Project Name/Location/ Status(a)	Design Capacity(b)	Distance (One-Way)	Container Technology Description					
			Vendor	Size	Volume	Light Weight	Commodity Weight	Commodity to Payload Weight Ratio(d)
	$STPD_5$	miles	Name	ft x ft(c)	yd^3	ST	ST	ST/ST
Bamberg/FRG/OP	300 252 actual avg	28	[illegible]	8'ø x 30'L	69.7	6.05 (38 m^3)	21.03 avg 24.24 gtd (28.40 max)	0.777 (0.814 max)
Kempten I/FRG/SD(e)	136	22	Altvater (Thyssen)	8.20'ø x 51.17'L	95.4 (73 m^3)	11.02	33.91	0.755
Kempten II/FRG/OP	136	22	Altvater (Thyssen)	8.65'ø x 64.72'L	130.7 (100 m^3)	16.50	49.60	0.750
Schwandorf/FRG/OP	1,600 present (2,000 future) 1,370 avg actual	64 max(f)	[illegible]	8.20'ø x 30.54'L	55.8 (427 m^3)	6.06	25.96 avg	0.811 avg
Troisdorf/FRG/OP	992 775 actual avg	46	[illegible]	8'x8'x30'L	71.1	6.60	21.53 actual avg	0.765
Winterthur/CH/OP	106.8 actual (165 max)	26	Multi-Lift	8.20'x8.20'x 18.7'L	46.5 (35 m^3)	5.50(g)	9.05 avg	0.612

Notes:
(a) O = operational; SD = shut down
(b) $STPD_5$ = short tons per day on 5-day a week basis
(c) Dimension of circle or rectangle
(d) Commodity to Payload Weight Ratio = Commodity weight ÷ payload
(e) Upgrading from $73m^3$ to $100m^3$ containers became necessary because of compaction problems
(f) Involves 3 different routes
(g) Includes turntables

Table 2. COMPARISON OF DESIGN PARAMETERS FOR MSW RAIL HAUL CONTAINER SYSTEMS OPERATING IN EUROPE - Rail Car Technology

Project	Rail Car Description						
Name/Location/Status(a)	Container & Car Configuration(b)	Overall Length	Specific Commodity Length(c)	Light Weight(d)	Gross Weight(e)	Commodity to Car Weight Ratio(f)	Rolling Resistance(g)
	#/shape/#	ft	ST/ft	ST	ST	ST/ST	Lb/ST
Bamberg/FRG/OP	2 CY/4 ax	64.29	0.719	22.10	80.40	0.575	4.184
Kempten I/FRG/SD	1 CY/4 ax	55.00	0.617	17.00	62.01	0.547	4.617
Kempten II/FRG/OP	1 CY/4 ax	72.16	0.687	19.04	85.93	0.577	4.070
Schwandorf/FRG/OP	2 CY/4 ax	65.37	0.795	30.94	84.98	0.611	4.101
Troisdorf/FRG/OP	2 SQ/4 ax	64.29	0.753	22.30	83.90	0.577	4.120
Winterthur/CH/OP	3 SQ/4 ax	64.29	0.451	25.90	71.41	0.406	4.382

Notes:
(a) O = operational; SD = shut down
(b) CY = cylinder; SQ = box shape
# of containers per car or number of axles per car
(c) Specific Commodity Length = Commodity Weight ÷ Overall Car Length
(d) Only rail car without any payload
(e) Weight of car loaded with full containers. See Table 1 for reference.
(f) Specific Commodity Weight = Commodity Weight ÷ Gross Weight of Car Loaded.
(g) Rolling resistance using Davis formula and 40 mph average speed.

Table 3. COMPARISON OF OPERATING PARAMETERS FOR MSW RAIL HAUL CONTAINER SYSTEMS OPERATING IN EUROPE

Project	Comm.	Load	Tonnage	Typical Train Configuration				Inventory of Rolling Stock(c)		Personnel Productivity
				Containers	Cars	Length	Full Weight	Containers	Cars	Overall
Name/Location/ Status	Year	Type(a)	STPY(b)	#	#	ft	ST	#	#	STPY/ Person
Bamberg/FRG/ OP	1980	MD	65,600	12	6	385	482	40	8	11,900
Kempten I/FRG/ SD	1979	MD	33,000	1	3	N.A.	N.A.	4	4	13,200
Kempten II/ FRG/OP	1981	MD	33,000	1	3	216	258	5	5	13,200
Schwandorf/ FRG/OP	1982	HD	356,300	2	varies(d)	varies(d)	varies(d)	152	38	12,700(e)
Troisdorf/ FRG/OP	1987	MD	201,500	2	18	1,157	1,518	85	21	10,300
Winterthur/ FRG/OP	1985	LD	28,232	3	4	286	257	34	6	8,100

Notes:
(a) HD = high density; MD = medium density; LD = low density
(b) STPY = short tons per year
(c) Includes varying number of spares.
(d) Three different trains are involved, only two of which are unit trains.
(e) Will increase with plant expansion.

Table 4. TRAIN DESIGN CHARACTERISTICS FOR 2-WAY CONTAINER RAIL HAUL SYSTEM TO SERVE RPP(a)

Rail Car Type	Container Name & Size	Car	[illegible] Commodity Load	Containers	Cars	Axles	Train Length	Refuse 6-Trains		Ash 6-Trains		Annual Round Trip Ton-Mileage
								Full Weight incl. 2,000 ST Commodity	Avg Gross Axle Weight	Full Weight incl. 600 ST Commodity	Avg Gross Axle Weight	
			(ST/container)	(#)	(#)	(#)	(ft)	(ST)	(ST/Axle)	(ST)	(ST/Axle)	(Ton-Miles)
I. European Conventional Design												
Std. Flat Car Std. Container	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Special Frame Car Mechanical Containers	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Std. Flat Car Std. Container	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Std. Flat Car Std. Container	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Std. Flat Car [illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
II. American Super Design												
Std. Flat Car Std. Container	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Std. Flat Car Std. Container	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Stand Alone Twin Stack [illegible] Std. Cont.	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Articulated Car [illegible] Std. Container	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Articulated Car [illegible] Ultra-light Containers	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Articulated Car [illegible] Std. Container	[illegible]		[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Std. Flat Car [illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]

Notes:
(a) One-way mileage = 22 miles door to door
Normal operating schedule: 52 weeks x 6 days = 312 days per year
Annual one-way train mileage: 52 x 6 x 22 = 6,864 miles/year
Daily refuse tonnage, or commodity load: R = 2,000 STPD$_6$
Daily ash tonnage, or commodity load: A = 600 STPD$_6$, i.e. an ash rate of 30% after ferrous scrap removal.
(b) Not always a perfect match between containers and car capacity. Depending on the degree of matching, either unit may have some excess capacity.
(c) HD = High Density; MD = Medium Density; LD = Low Density - depends on refuse composition and compaction rates in transfer station
(d) SD = Standard Container
(e) TS/SA = Twin Stack/Stand Alone
(f) TS/A = Twin Stack/Articulated
(g) UL = Ultra-light Container

Economical Disposal of Municipal Solid Wastes With Minimal Discharges Into the Atmosphere and Maximum Recovery of Energy Values

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ABSTRACT

A process has been developed at Brookhaven National Laboratory for the treatment and disposal of municipal solid wastes with minimal discharges into the atmosphere and maximum recovery of the energy values still contained in the wastes. These energy values are recovered as zero ash, zero sulfur, zero nitrogen particulate carbon (carbon black) fuels and hydrogen-rich and/or methane-rich gaseous co-products.

The process consists of hydrogasifying the MSW (or any carbonaceous feedstock) to form a methane-rich process gas, which is then thermally decomposed (cracked) to form the primary product, carbon black, and hydrogen which is recycled to the hydrogasifier. Oxygen in the MSW is removed as water from the hydrogasifer effluent before it enters the methane decomposer. Any remaining hydrogen in the MSW feed is ultimately removed from the process as a co-product gas. Chlorine in feed containing PVCs, for example, is collected as highly dilute hydrochloric acid in the condensed water discharged from the recuperative partial condenser. Desulfurization is not required to produce sulfur-free carbon black *per se*. Various options are available for desulfurization of the co-product gas or gases. Since the process operates under a highly reducing hydrogen atmosphere, toxic oxygenated compounds such as dioxins cannot be discharged.

Conversion of carbon in the MSW to carbon black and combustible carbon compounds in the co-product gas or gases is almost quantitative. Process thermal efficiencies are of the order of 90% or better. Consequently, the proposed process for the disposal of MSW is believed to be considerably more attractive, both environmentally and economically, than previously available processes.

PLANNING FOR RESOURCE RECOVERY FACILITIES UNDER NEW STATE AND FEDERAL REGULATIONS

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Sheree L. Stewart
John J. Trotter
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ABSTRACT

A resource recovery facility is perhaps the single most complex public benefit project ever attempted by local government. What is somewhat unique about resource recovery projects, aside from their complexity, is the tortuous paths they often must take from project inception through construction. An extremely important aspect of assuring success for such complex projects is the certainty of the rules and regulations governing the permitting of these projects. Decisions made at the federal and state level impact the planning and decision making process of resource recovery projects at the local level. This paper will first discuss the general planning process for resource recovery projects and then examine how changing federal and state regulations in the areas of air emissions and ash residue management significantly affect the economics and structure of the planning process.

INTRODUCTION

How to dispose of the cans, cereal boxes, newspapers, tires, bottles, and other castoffs of America's communities in an environmentally sound and economically efficient way has become a problem of critical proportions. Counties and cities across the United States are being confronted with the ever increasing volume of solid waste generated by our society's residential, commercial and industrial activities. It has been estimated by the U.S. Environmental Protection Agency (EPA) that each person in the United States generated 3.6 pounds of solid waste per day in 1986 on a nationwide basis (1). However, many states currently report much higher per capita generation rates suggesting that current EPA estimates seriously underestimate the magnitude of the nation's solid waste disposal problem. With one-third of the nation's sanitary landfills expected to be full by 1993, and the per capita generation rate increasing in many communities, the search for alternative long-term solutions to solid waste disposal has lead many communities to develop an "integrated waste management" approach.

The term "integrated waste management" refers to a solid waste management technique which allows the use of a variety of management practices to handle a community's solid waste stream in an economic manner that is protective of human health, safety, and the environment (2). Such an integrated system incorporates the following major components:

- Source Reduction
- Recycling and Reuse of Materials
- Waste Incineration
- Sanitary Landfilling

Solid waste management plans now being developed by communities across the United States are emphasizing a mix of these different management elements. This allows each community to design a management system which takes into account local socioeconomic and environmental conditions. For many communities, implementation of resource recovery facilities are an integral part of their solid waste management plans (Figure 1). While source reduction and recycling efforts help extend the useful life of existing facilities, they cannot by themselves alleviate the need for the development of additional long-term solid waste disposal capacity in these communities. Unstable market conditions for the sale of recyclable materials and the lack of available land for future landfill expansion have prompted many communities consider resource recovery facilities as a major component of their long-term solid waste management and disposal system.

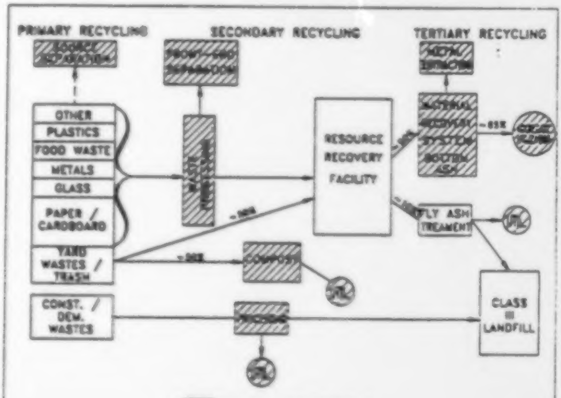
PRIMARY RECYCLING
SECONDARY RECYCLING
TERTIARY RECYCLING
SOURCE SEPARATION
FRONT-END SEPARATION
METAL EXTRACTION
OTHER
PLASTICS
FOOD WASTE
METALS
GLASS
PAPER / CARDBOARD
YARD WASTES / TRASH
CONST. / DEM. WASTES
RESOURCE RECOVERY FACILITY
MATERIAL RECOVERY SYSTEM
BOTTOM ASH
FLY ASH TREAMENT
COMPOST
CLASS III LANDFILL
~85%
(▨ RECYCLING COMPONENTS)

Figure 1. Integrated Waste Management

PLANNING OF A RESOURCE RECOVERY PROJECT

Planning and implementation of a resource recovery project is a lengthy complex process which consists of several phases requiring "go/no-go" decisions to be made by the community (3). Since no two projects are identical, the discussion on project planning is by nature generic in approach. Table 1 displays, however, the major planning steps normally followed for a resource recovery project. These steps can be modified by a community taking into account its individual concerns and needs.

PHASE I - FEASIBILITY ANALYSIS

This phase is preceded by a preliminary, and somewhat informal, investigation by a community or project sponsor to insure that the necessary project elements, namely the shortage of available landfill space, the availability of a potential energy market, and relatively sharp increases in projected disposal costs exist, and make further study of a potential resource recovery project reasonable.

During Phase I, the feasibility of such a project is evaluated in detail. An analysis is undertaken of the community's existing and projected waste stream to determine the competition of the waste stream and the ultimate size of the facility. Markets and technologies are examined to determine

TABLE 1 TYPICAL IMPLEMENTATION PHASES OF A RESOURCE RECOVERY PROJECT

Phase I - Feasibility Analysis

- Analyze Waste Stream
- Analyze Waste Disposal Practices
- Analyze Energy and Materials Market
- Analyze Feasible Waste-to-Energy Technologies
- Analyze Potential Facility Sites
- Review Permitting Requirements
- Analyze Ash Management Alternatives
- Risk and Legal Assessment
- Conduct Financial Analysis
- Develop Project Alternatives
- Go/No-Go Decision

Phase II - Procurement

- Select Project Alternative
- Select Site and Acquire
- Obtain Permit
- Negotiate Market Contracts
- Negotiate Waste Stream Guarantee
- Develop Financing Plan
- Issue RFQ/RFP
- Select Contractor
- Negotiate Operating Contract
- Issue Notice-To-Proceed

Phase III - Plant Construction

- Site Preparation
- Complete Final Design
- Equipment Ordered
- Building Constructed
- Equipment Installed
- Testing and Startup
- Acceptance Testing
- Certificate of Completion

Phase IV - Plant Operations

- Service Fee Payment
- Annual Tipping Fee Adjustment
- Annual Report (Optional)
- Facility Retesting (Optional)

whether the energy and/or materials recovered by this system can generate adequate revenues to offset the construction, operating, and financing costs for the facility. Feasible sites are investigated along with an analysis of the technical, environmental, and regulatory requirements on the federal and state level for permitting a facility. In addition, a risk assessment is undertaken to help determine the risk posture to be taken by the community with regards to waste-to-energy technology, project ownership, operation, and financing. A formal feasibility report, which is presented to the implementing entity, usually documents all activities in this phase. Phase I ends with a "go/no-go" decision on the part of the community to either proceed with implementation of the project or to terminate activities for the foreseeable future.

PHASE II - PROCUREMENT

Phase II incorporates all the steps necessary to procure the waste-to-energy system desired by the community including: contracts for waste supply, energy and materials markets, and plant construction and operation; acquiring the project site; obtaining all environmental permits and/or regulatory approvals; and securing the financing for the project. Phase II usually requires the development of specialized procurement documents such as a Request-for-Qualifications (RFQ) and Request-for-Proposal (RFP). Dependent on the financing approach selected by the community, documents such as a Bond or Trust Indenture, Engineer's Feasibility Report, and Bond Prospectus may need to be written.

PHASE III - PLANT CONSTRUCTION

Phase III covers all the steps necessary in the construction of the waste-to-energy facility including: site preparation; final design; ordering facility equipment; installing this equipment; testing and startup of the facility; and completing acceptance tests. These tasks are usually undertaken by the selected facility contractor, although most communities obtain the services of a knowledgeable engineering consulting firm to independently monitor construction activities and the facility's acceptance tests. Phase III usually ends with the facility successfully completing the contractual acceptance test procedures and the community issuing a certificate of completion to the contractor.

PHASE IV - PLANT OPERATIONS

Phase IV covers the period from plant acceptance through the length of the operations period, which may last 20 years or more in the case of a full-service contractor for a publicly or privately-owned facility. These contracts usually have clauses allowing them to be renegotiated after this time period.

PROJECT SCHEDULING

The duration of the implementation process for a resource recovery project depends upon a variety of factors which the sponsoring community may or may not control. Factors such as delays in regulatory reviews, procurement approach and project siting, for example, impact the length of time which a community may need to conclude the implementation of a resource recovery project. With significant numbers of resource recovery projects being planned and implemented in recent years, however, the procurement, financing, and contractual procedures for these projects have been more standardized, thereby reducing the length of time required to implement such projects. Early projects required upwards of two to three years to complete planning and procurement activities, with an additional two years to complete all contractual negotiations and arrange for financing. More recent projects have completed these Phase I and Phase II activities within two to three years.

Unfortunately, the time process for permitting such projects has neither been reduced nor simplified in recent years. As resource recovery projects move toward implementation, they come under increasing fire from citizens and environmental activists concerned with environmental impacts such as air quality and ash residue management. Permit hearings for such projects have now become protracted struggles across the United States between these groups and the project sponsor. This increased public attention and controversy has resulted in both federal and state agencies re-evaluating their standards currently on the books for resource recovery projects. The balance of this paper will discuss the impact of these changing regulations upon the planning and implementation of resource recovery projects.

RECENT REGULATORY ACTIVITY

Long before construction activities for a resource recovery facility can begin, project participants must obtain all the necessary federal, state and local regulatory approvals for construction and operation. The Federal Clean Air Act and the regulations promulgated by the EPA form the legal basis for permitting air emissions from resource recovery facilities.

There has been a concern, however, voiced by many critics that the current Federal air regulations have become outdated based upon current knowledge about air emissions and ash residue disposal and their possible impacts upon human health and safety. Much of the existing Federal New Source Performance Standards were promulgated, for example, in an era where smoke and soot from municipal incinerators were key citizen concerns. Action to update these air regulations has been proposed by both the Congress and the EPA.

Several pieces of legislation were introduced in the 100th Congress regarding the permitting and operations of resource recovery facilities. The following subsections briefly describe legislation proposed by several Congressmen and Senators which would require the EPA to set stringent emissions and ash residue handling standards for resource recovery facilities.

FLORIO LEGISLATION

Legislation introduced by Congressman Florio (NJ) in the last session of Congress was designed to amend the Clean Air Act to control emissions of some two dozen air pollutants and ash residues from municipal waste incinerators. The legislation would have required that the EPA promulgate new standards which reflected the "greatest degree of emission limitation achievable through application of best available control technologies." These technologies could either have been: (1) achieved in practice by municipal waste combustion units; or (2) contained in the implementation plan of any State, unless the owner or operator can demonstrate that such performance is not achievable. The term "available technologies" is defined as: electrostatic precipitators, fabric filtration, flue gas scrubbers (wet and dry), spray dry scrubbers, condensation scrubbers, catalytic reduction, wet flue gas denitrification, selective non-catalytic reduction, and catalytic oxidation.

The legislation would have required the EPA to promulgate strict rules mandating that owners or operators of such facilities continuously monitor emissions for some nine pollutants and combustion factors, and for all emissions not subject to standards, periodic monitoring every six months. Additionally, each community permitting a resource recovery facility would be required to develop a solid waste management plan which could be subject to citizen lawsuits if the community acted in violation of law in developing its plan. With respect to existing resource recovery facilities, the legislation would have required EPA to promulgate new standards and regulations requiring such units to comply with emission standards in effect. The legislation included a retrofit timetable based upon costs providing for compliance with such standards.

Also, the Florio legislation would have required disposal of combined ash in a Subtitle C landfill. Requirements with regard to testing were stringent; quarterly sampling of fly and bottom ash would be required to reflect leaching characteristics, total chemical analysis, respirability and toxicity.

DURENBERGER LEGISLATION

Companion Senate legislation was also introduced in the last session of Congress by Senator David Durenberger (MN). Although similar to Congressman Florio's bill, this legislation was more restrictive in its regulation of air emissions and less restrictive in regards to ash management. Both pieces of legislation failed to be enacted primarily due to the controversy surrounding the re-authorization of the Clean Air Act. It is anticipated by most Congressional observers that similar legislation will be re-introduced in the 101st Congress.

LUKEN LEGISLATION

Public concern about the controversial voyage of the infamous garbage barge, Mobro 4000, prompted Congressman Luken's (OH) Subcommittee on Transportation, Tourism and Hazardous Materials to hold numerous hearings on solid and hazardous waste handling during the 100th Congress. An outgrowth of these hearings was the introduction of legislation by Congressmen Luken and Wittaker (CA) giving EPA the authority to regulate the disposal of ash from resource recovery facilities.

The Luken bill would have required EPA to develop a new toxicity test for ash and to promulgate regulations for handling and disposal of ash. Further, it would have required states to adopt and implement a program to ensure compliance with federal regulations and would have given EPA enforcement power in states that failed to do so.

EPA ACTIONS

In June 1987, EPA formally submitted its long awaited report to Congress detailing the results of a three year research effort on municipal incineration. EPA later published a notice in the Federal Register describing the results of their preliminary assessment of air emissions from resource recovery facilities (4).

In it, the EPA concluded that the air emissions from municipal waste combustors may contribute to the endangerment of public health and welfare through increased risk from cancer and other illnesses. The agency revealed also that it had made a determination to promulgate new standards for one or more pollutants not previously regulated as non-criteria pollutants. In deciding this approach, EPA invoked Section 111(d) of the Clean Air Act which requires the issuance of more stringent New Source Performance Standards (NSPS) for future resource recovery facilities followed by the development by the states of specific emission standards for existing plants.

By taking this approach, the EPA denied the petitions filed by the National Resource Defense Council (NRDC) and the States of New York, Connecticut, and Rhode Island to regulate the air emissions of resource recovery plants under Section 112 of the Clean Air Act. Currently, with the exception of certain trace metals, none of the emissions from these plants are listed as hazardous pollutants under Section 112. In addition, EPA argued that Section 111(d) plans for existing plants would be more flexible since they could consider on a case-by-case basis such relevant factors as the remaining useful lives of these facilities and the cost of retrofitting emission controls.

Section III of the Clean Air Act requires each state to establish plans, similar to State Implementation Plans (SIP), to address existing sources of air pollution through performance standards. These standards would then reflect the degree of emission limits achievable through the use of "Best Available Control Technology" (BACT). That is, the best technology for achieving the maximum degree of air emissions reduction. Over the years, however, the EPA and the courts have interpreted the evaluation of BACT as requiring an analysis of the energy, environmental, and economic impacts associated with each alternative technology before one type of pollution control technology or process is selected over another.

The seldom used Section 111(d) authority over existing facilities, however, is extremely controversial and may be subject to litigation in the future before it is implemented by the States. In contrast, new facilities will immediately feel the impact of these proposed policy changes. Recent EPA permit remands and the policy guidance document issued by EPA to state and local PSD permitting agencies indicate that all municipal waste incineration projects will probably be mandated to have the following equipment (or capability) at a minimum:

- Dry scrubbers;
- Electrostatic Precipitators or Fabric Filters;
- Auxiliary Burners for Start-up and Combustion Control; and
- Combustion Temperature Maintenance Conditions at or above 1800 degrees F.

In some locations where NOx emissions are major air quality concerns, such facilities may require incorporation of de-NOx or catalytic conversion equipment to control NOx emissions. It is also possible that BACT determinations will vary according to type of incineration facility (e.g., mass-burn, starved-air, and RDF).

Unfortunately, the recent EPA actions fall short of providing local governments planning resource recovery facilities with complete regulatory certainty. There will be risk with respect to changes in environmental laws. The proposed time schedule for future EPA actions (Table 2) has come in for considerable criticism by those who question whether their policy guidance will be taken seriously by local governments on the brink of investing in multi-million dollar projects.

TABLE 2. PROPOSED EPA ACTIONS REGARDING NEW SOURCE PERFORMANCE STANDARDS

Action Item	Date
Proposal of New Source Performance Standards (new facilities)	November 1989
Issuance of Draft Guidelines (existing facilities)	November 1989
Promulgation of New Source Performance Standards (new facilities)	December 1990
Issuance of Final Guidelines (existing facilities)	December 1990
State Plan (existing facilities)	September 1991

IMPACTS OF PROPOSED REGULATORY CHANGES UPON PLANNING EFFORTS

As indicated in the previous discussion, there has been a considerable amount of technical, legal, and political debate regarding changes in environmental regulations over the past several years for resource recovery facilities. This debate appears now to be directed towards promulgating new federal and state standards and rules which severely restrict the emissions from resource recovery facilities and treat ash residues from such facilities not unlike that of hazardous waste. It is beyond the scope of this paper to examine whether this trend makes good public policy sense, especially when these combustion sources emit relatively insignificant fractions of the total point source contributions in most local communities across the United States. As shown in Table 3, for example, the total SO_2 emitted by resource recovery facilities in Hillsborough County, Florida (Tampa area) during 1985 was only 0.1 percent of the total County atmospheric load of SO_2. In comparison, a newly constructed coal fired power plant released nearly three quarters of the entire amount of SO_2 emitted by all major point sources in the County combined. In total, all coal-fired power plants together emitted over 95 percent of this total County's load (5).

This result is not surprising since solid waste is an extremely low-sulfur fuel, generally containing less than 0.2 percent sulfur as compared to 3.0 percent for most low-sulfur western coals used by U.S. electric utilities. Consequently, the incineration of solid waste to produce electric energy should have a beneficial effect on the environment in the form of lower SO_2 and acid gas emissions as compared to that produced by coal or oil-fired, base load power plants.

TABLE 3. SO_2 EMISSIONS IN HILLSBOROUGH COUNTY BY SOURCES, 1985

POINT SOURCE	TONS/YEAR SO_2	PERCENT
Electrical Generation	197,750.0	95.1
Industrial Fuel Combustion	2,286.1	1.1
Phosphate Production	7,655.7	3.7
Solid Waste Disposal	210.0	0.1
TOTAL	201,901.8	

Notwithstanding these arguments, the current trend is towards increased regulations on the state and federal level for resource recovery facilities. As indicated previously, the current state of regulatory flux on air emissions and ash disposal provides an uncertain environment for communities planning such facilities. Since the overall cost to the homeowner or solid waste ratepayer is a key concern to political decision-makers of most communities planning these projects, there is always a balancing of project economics against the installation of alternative pollution control devices offering reduced plant emissions.

Unfortunately, those devices offering the lowest costs may result in the highest risks to the community, especially if federal and state regulations and standards are promulgated which require that such plants meet these new standards or shut down. Many communities and plant vendors are opting for the more expensive pollution control device, regardless of price, because they are fearful of plant retrofits soon after these plants are in commercial operation.

Table 4 illustrates the type of data that can be developed to compare the economic consequences of alternative particulate, SO_2, and acid gas control strategies for resource recovery facilities. This example was prepared for a 1,000 ton per day mass burn resource recovery facility to be located in Spokane, Washington (6).

The data presented include the total annual cost for each control system, the annual cost per ton of solid waste processed, and the differential of each of the alternative's cost over the base capital, O&M, and total costs. The annual cost of emission controls per ton of refuse is meaningful, in that it represents the relative increase in cost per ton of refuse disposed of at the facility that is directly attributable to the addition of control equipment.

Existing operations require an entirely different approach in order to estimate the cost of installing advanced air pollution control equipment. Demolition and reconstruction of existing and new equipment will undoubtedly impact the existing operations of such facilities which must be reflected in the overall tipping fees for users of the facility. To illustrate the cost to users of such facilities, order of magnitude estimates to complete a plant retrofit were made of two facilities located in Hillsborough County, Florida. Table 5 illustrates the results of this study.

Based upon the preliminary reports prepared by each plant operator on construction costs, the economic impact of the installation of acid gas scrubbers on each community's tipping fees were calculated. Some of these cost parameters are based on current operating experience for these facilities (i.e., energy revenues, energy production, builder's risk insurance premium, pass through costs, etc.), while others are based upon current contractual arrangements between the plant operator and owner (i.e., share of energy revenues, service fees, etc.). Estimates of construction costs, the length of individual unit shutdown, and increases in the service fee to accommodate the addition of scrubbing equipment were based upon the preliminary analysis undertaken by the two plant operators.

TABLE 4. SPOKANE REGIONAL WASTE TO ENERGY PROJECT ECONOMIC COMPARISON OF ALTERNATIVE PARTICULATE, SO_2, AND ACID GAS CONTROL STRATEGIES

	Particulate Control			SO_2 & Acid Gas Control			Additional NO_x Control	
	ESP	Fabric Filter	Venturi Scrubber	ESP & Wet Scrubber	Dry Scrubber & ESP	Dry Scrubber & Filter	Selective Catalytic Reduction	Thermal deNOx
CAPITAL COST								
Inlet ACFM (X 1,000)	265	265	265	265	265	265	265	265
Approximate Capital Cost per ACFM	9.81	9.24	5.00	18.87	21.13	22.64	16.98	7.55
Total Capital Cost ($1,000)	2,600	2,450	1,325	5,000	5,600	6,000	4,500	2,000
Net Debt Service ($1,000/yr)	225	212	115	432	486	519	389	173
Cost per Ton Refuse ($)	0.62	0.58	0.31	1.18	1.33	1.42	1.07	0.47
Differential ($)	0.31	0.27	0.00	0.87	1.01	1.11	-	-
O & M COST ($1,000/yr, 1986 $)								
Electricity (@ 40 mils/kwh)	52.6	56.5	152.1	105.6	91.4	95.3	24.6	85.5
Labor	50	50	100	150	100	100	50	50
Materials and Supplies	52.0	49.0	132.5	500.0	280.0	300.0	1,124.9	60.0
Chemicals	0.0	0.0	0.0	45.5	19.1	19.1	13.9	
Water	0.0	0.0	394.2	394.2	157.7	157.7	0.0	0.0
Ash/Sludge Disposal	101.0	105.3	121.2	222.3	206.2	206.2	25.0	
Bag Replacement	0	224	0	0	0	66.3	0	0
Total 1986 O & M Cost	255.6	484.7	900.0	1,417.6	854.4	944.5	1,238.4	214.4
Cost per Ton Refuse ($)	0.70	1.33	2.47	3.88	2.36	2.59	3.39	0.59
Differential ($)	0.00	0.63	1.77	3.16	1.66	1.89	-	-
TOTAL 1986 ANNUAL COST ($1,000)	480	697	1,015	1,850	1,330	1,463	1,628	387
Differential	0.0	216.2	534.1	1,369.5	850.3	983.0	-	-
TOTAL 1986 ANNUAL COST ($/Ton Refuse)	1.32	1.91	2.78	5.07	3.67	4.01	4.46	1.06
Differential	0.00	0.59	1.46	3.75	2.35	2.69	-	-

Two scenarios were developed to compare the economic consequences of the addition of acid gas scrubbing equipment to these three facilities. The first scenario (Option A) assumes that the scrubber units would be installed sequentially so that each boiler train would have to be taken off-line to accomplish the tie-in of the acid gas scrubbing equipment. Downtime of each unit was anticipated to be eight months during which there would be loss of energy revenues and increased costs of landfilling to accommodate the bypassed solid waste quantities. The capital funding costs of construction over this time period would have to be added to the lost revenues. The second scenario (Option B) makes the assumption that the installation of the acid gas scrubbers can be accomplished at the time of normal boiler train maintenance such that each unit would not incur additional downtime and the associated loss of revenues. The costs associated with this alternative would essentially be the capital costs of construction.

TABLE 5. Increases in Disposal Costs (Tipping Fees) For Emission Control Retrofit of Tampa Bay Resource Recovery Facilities (1987 $ Per Ton)

City of Tampa McKay Bay Facility:	Option A	Option B
Capital Financing Needs	$ 6.36	$ 7.63
Revenue Loss and Extra Expense	47.75	--
Total Tipping Fee Increase	$ 54.10	$ 7.63
Current Tipping Fee	$ 55.77	$ 55.77
Resultant Tipping Fee	$ 109.87	$ 63.40

Hillsborough County Resource Recovery Facility:	Option A	Option B
Capital Financing Needs	$ 6.41	$ 6.98
Revenue Loss and Extra Expense	72.58	--
Total Tipping Fee Increase	$ 78.99	$ 6.98
Current Tipping Fee	$ 59.75	$ 59.75
Resultant Tipping Fee	$ 138.74	$ 66.73

ASH RESIDUE MANAGEMENT

The lack of a comprehensive federal regulation program for ash management has been due in part to Congress' inability to pass legislation giving EPA the proper authority to promulgate regulations. Although EPA has issued a guidance document (7) with preliminary recommendations, the uncertainties regarding federal ash regulations will likely continue for some time. How then do communities plan for their facilities in this regulatory vacuum in the interim?

At the present time, regulations allow disposal of combined ash from a residential and commercial MSW incinerator in a Class I landfill. There has been considerable debate recently on whether ash is subject to testing and which sampling and analysis method is most appropriate to determine whether the material is hazardous or non-hazardous. Under the Resource Conservation and Recovery Act (RCRA), if a material does not "pass" the EP Toxicity test, it is considered a hazardous waste. EPA is currently working to develop a new test that is more appropriate and representative of actual field conditions.

EPA's guidance for ash management provides very conservative management recommendations. The guidance document provides transitional recommendations for operating facilities and general guidance for proper handling, transport, and design of disposal units. The recommendations provide extensive environmental controls. EPA's recommendations regarding design of disposal units are dependent upon whether the ash is monofilled or co-disposed with MSW in a landfill. The recommended liner specifications are shown on Figure 2.

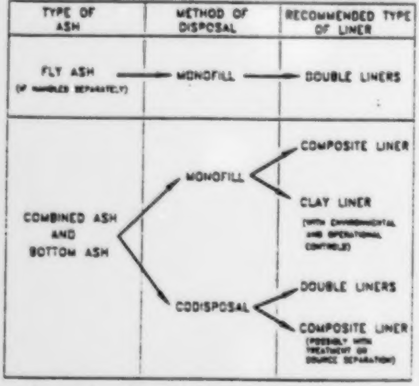
TYPE OF ASH
METHOD OF DISPOSAL
RECOMMENDED TYPE OF LINER
FLY ASH
(IF HANDLED SEPARATELY)
MONOFILL
DOUBLE LINERS
COMBINED ASH AND BOTTOM ASH
MONOFILL
COMPOSITE LINER
CLAY LINER
(WITH ENVIRONMENTAL AND OPERATIONAL CONTROLS)
CODISPOSAL
DOUBLE LINERS
COMPOSITE LINER
(POSSIBLY WITH TREATMENT OR SOURCE SEPARATION)

Figure 2. EPA's Current Recommendactions for Liner Requirements for Ash Disposal Units

Fly ash and bottom ash can be disposed of separately or combined. Most facilities generally combine the ash streams internally and manage it as one stream. EPA is currently recommending that if a facility manages fly ash separately, it should be disposed of in a monofill with a double-lined/leachate collection system. It should not be co-disposed with MSW. Bottom ash or combined ash, according to EPA, should be disposed of in a double-lined/leachate collection disposal unit if it is co-disposed with MSW. Bottom ash or combined ash alone should be disposed of in a composite-lined landfill or clay liner. Several states are following EPA's recommendations in recommending monofills.

EPA is also examining the issue of ash treatment and utilization as an alternative to disposal. It is likely that future regulations will require that any proposal for alternative utilization methods include adequate testing to determine environmental compliance.

The alternatives that are currently available for ash management are shown in Figure 3. The most appropriate option for disposal or treatment and utilization will obviously vary from facility to facility depending on its geographical area, operational size, and specific ash characteristics. The selection of an appropriate ash management option is based on several very important factors, which can be summarized as follows: (a) the specific physical and chemical characteristics of the ash, (b) the technology for treatment, (c) the availability of potential markets, (d) local environmental effects of various disposal and reuse options, and (e) treatment or disposal costs.

ASH DISPOSAL

Disposal of ash in a double-lined monofill or landfill is very expensive, but effective in terms of protecting the environment and human health. Data from a recent study (8) indicates actual leachate from monofill sites do not exceed maximum allowable toxicity limits for any constituent. Nearby groundwater wells produce groundwater that passes Primary and Secondary Drinking Water Standards. A recent health risk assessment (9) for ash disposal at a monofill quantitatively calculated risks associated with fugitive emissions and groundwater contamination. Conclusions drawn from this study indicate the following: (a) the primary potential for human exposure appears to be via emissions to the air not groundwater, (b) emissions are a function of moisture content, management practices, and physical characteristics of the ash after disposal, and (c) the risks to human health are extremely low for ash disposal at a monofill.

The design features of a double-lined ash monofill are numerous and the construction and operation is very expensive. Double-lined composite monofills generally cost in the range of $150,000 to $200,000 per acre. A schematic diagram of the design of a monofill is presented in Figure 4. Double liner systems consist of a protective clay liner, two flexible membrane liners, two leachate collection systems, and cover soil to protect the liners. Figure 4 shows a detailed cross-section of the basic components of the composite double liner system. The composite double liner system consists of a primary leachate collection and removal system (either synthetic drainage

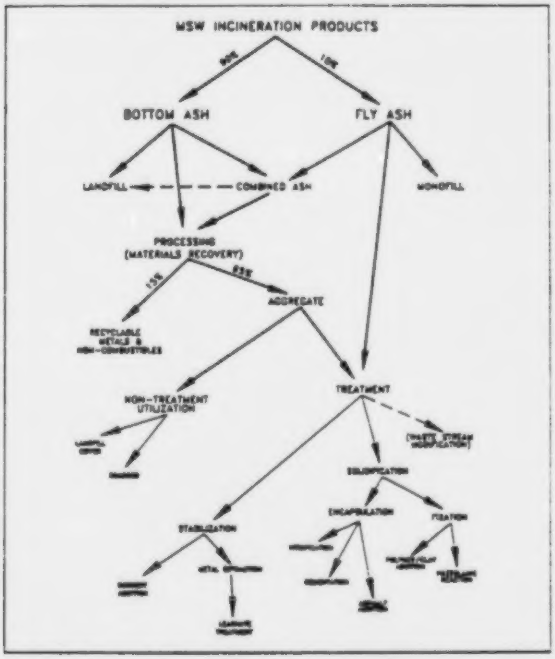
MSW INCINERATION PRODUCTS
90%
10%
BOTTOM ASH
FLY ASH
LANDFILL
COMBINED ASH
MONOFILL
PROCESSING (MATERIALS RECOVERY)
15%
85%
AGGREGATE
RECYCLABLE METALS & NON-COMBUSTIBLES
TREATMENT
NON-TREATMENT UTILIZATION
(WASTE STREAM MODIFICATION)
SOLIDIFICATION
ENCAPSULATION
FIXATION
STABILIZATION
LEACHATE TREATMENT

Figure 3. MSW Incineration Products Management Alternatives

netting or PVC pipes and sand), a top flexible membrane liner, a secondary leachate collection system, a bottom composite flexible membrane, and a low permeability soil liner.

At the present time, siting and construction of an ash monofill, as part of an integrated solid waste management program, will minimize the regulatory risk assumed by a community. In most cases, it would be prudent for communities planning for an MSW incinerator facility at the present time to incorporate the siting and construction of a monofill into their solid waste plans. This conservative approach would allow a community to convert the monofill to handle other wastes in the future as ash utilization becomes an (environmentally) feasible alternative.

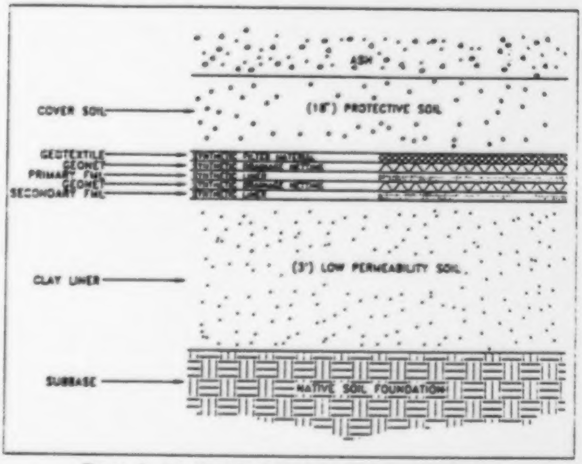

Figure 4. Ash Monofill Composite Double Liner System

ASH TREATMENT AND UTILIZATION ALTERNATIVES

The treatment alternatives for ash fall into three broad categories - solidification, stabilization and waste stream modification. It is important to note that unprocessed bottom ash or combined ash is very difficult to handle or treat. As shown in Figure 3, a materials recovery system can process and remove 15% of the entire ash stream as recyclable metals (and non-combustibles). After the ash is processed, the resulting product is a 3/8 or 3/4 inch or less aggregate product. It is this portion of the ash stream that can be potentially treated and re-used.

Solidification is generally considered a type of physical treatment involving one of three processes: encapsulation, vitrification or fixation. The most promising types of solidification treatments include cementation, an encapsulation process whereby the cement blocks may be utilized as artificial reef components, foundation or erosion control, and fixation involving the addition of polymers or clays. Stabilization is a treatment process that incorporates a chemical reaction, either by extracting metals via an acid wash or adding a sorbent to chemically bind the metals in the ash. Waste stream modification can be accomplished through front-end separation of components such as batteries. Front-end separation has been shown to minimally reduce heavy metals in the ash stream.

The costs associated with the various treatment alternatives range from $3 per ton to $65 per ton of ash treated. In general, the fixation (solidification) treatments and the sorbent addition (stabilization) treatments tend to be least expensive, in the $3 - $10 per ton range. The costs of producing cement blocks are in the range of $10 - $20 per ton. The most expensive treatment processes are vitrification, metal extraction, and some of the stabilization techniques involving expensive proprietary chemicals. These are in the range of $35 - $65 per ton of ash treated. This seems to be an extremely high treatment cost, but if it is applied to the fly ash component only, this cost, which represents approximately ten percent of the total ash stream, is quite reasonable. To illustrate this, a case analysis was constructed to reflect the impact of treatment costs to the tipping fee at a generic 1000 tpd resource recovery facility. The case study compares the relative increase in the tipping fee of the following treatment alternatives: (a) co-disposal of combined ash in a sanitary landfill (base case); (b) disposal of combined ash in a secure monofill; (c) stabilization of the combined ash stream; (d) treatment of the fly ash for metal extraction; and (e) stabilization/solidification of the fly ash stream. Figure 5 demonstrates that the increase in the tipping fee is approximately 5% for the most expensive treatment for fly ash. The projected tipping fee increases shown in Figure 5 should be considered representative, but nevertheless shows it would be prudent for a facility operator to consider ash treatment and utilization as an alternative to the disposal of ash.

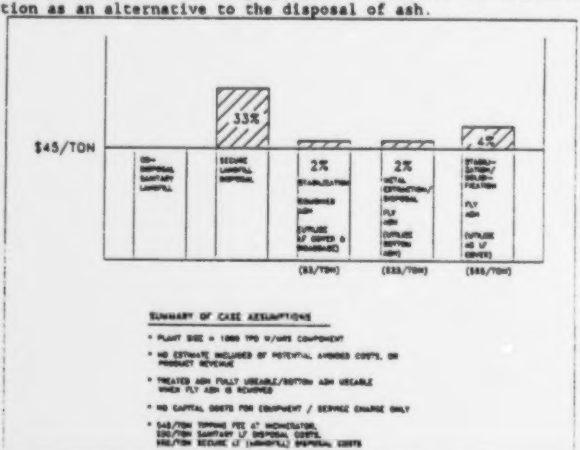

Figure 5. Estimated Percent Increase in Tipping Fee at Incinerator as a Result of Various Treatment Alternatives

There is extensive research underway at the present time by EPA, ASME, and private industry to identify safe utilization alternatives for ash aggregate. The most promising utilization alternatives currently include the use of untreated bottom ash or treated combined ash as aggregate for roadbase and admixture in concrete, cement blocks, or asphalt. Regardless of the particular utilization alternative to be used, the ash additive and/or post-treatment product will need to pass an EP-Tox or similar type test. The appropriate tests have yet to be determined by EPA. The potential for such uses depend somewhat on provisions incorporated into new regulations on the federal and state level that allow for safe utilization of ash.

CONCLUSIONS

The planning effort required by a community to successfully implement a resource recovery project has become a challenging responsibility for public officials and consultants alike. In the last few years, the permitting process for such facilities has been stymied by regulatory fragmentation, organized opposition groups espousing NIMBY philosophies, and outdated standards and rules. Even under the best of conditions and regulatory certainty, this effort is a time-consuming and tedious process. With regulatory uncertainty, however, efforts to plan and implement these essential public benefit projects can become tortuous and frustrating experiences for many communities.

The ill-fated voyage of the garbage barge Mobro did more to reawaken interest in solid waste than any other public event since Earth Day in 1970. Both Congress and EPA were prompted to rediscover solid waste disposal and put it back on the public agenda. While Congress failed to enact significant legislation in the 100th Congress, the numerous public hearings held on the Florio, Luken and Durenberger bills concerning air emissions and ash residue management has resulted in some movement towards regulatory certainty for resource recovery facilities. EPA Guidance documents on incinerator operation and performance and ash testing, utilization, and monofill construction, will assist the planning efforts of future resource recovery projects. What is missing from the equation is the certainty that installation of such "best available control technologies" will shield these communities from the cost of facility retrofit only a few years in operation.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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OVERCOMING PUBLIC CONCERNS SURROUNDING MSW INCINERATOR FACILITIES BY WORKING HARD ON THE "SOFT" POLITICAL ISSUES

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ABSTRACT

"Grappling with garbage". It sounds like the latest attraction at the local mud wrestling arena. Our society is almost literally awash in a sea of garbage which has created problems and challenges ranging from the search for the most effective technologies for disposal to accomodating the conflicting environmental philosophies regarding the technologies.

The public finds itself compelled to take a role beyond the weekly ritual of taking a garbage can out to the curb. The Waste Management Industry must deal directly with the public concerns regarding economic, social and environmental issues created by the development of an MSW incineration facility.

A new approach has to be taken by developers of MSW facilities. Although the problem of waste disposal continues to grow at an alarming rate, the "not in my back yard" philosophy has grown at an equally alarming rate.

This paper will explore the political issues surrounding the development of MSW facilities. Examples of success stories and failures will be discussed and techniques useful in dealing with political issues are also examined.

THE MEDIUM IS THE MESSAGE

"Grappling with garbage". It sounds like the latest attraction at the local mud wrestling arena. In fact, there is nothing attractive or funny about it. Our society is almost literally awash in a sea of garbage which has created problems and challenges ranging from the search for the most effective technologies for disposal to accomodating these conflicting environmental philosophies regarding the technologies.

As the spreading sea of refuse continues to expand, the public will find itself compelled to take a role beyond the weekly ritual of taking a garbage can out to the curb. Federal, state and local permitting no longer are the sole hurdles that facilities of this kind must overcome in order to get built. Industry must deal directly with the public concerns regarding economic, social and environmental issues created by the development of one of these facilities as well.

The evolution of the waste handling industry has been very rapid in the last decade. Certainly, the incineration branch of the industry has relied heavily on the engineering and scientific professions for the technical knowhow required to create a project. Until recently, it has been the technology that has driven projects forward, but that pattern has now been turned on its head. What was once considered the "fluff and stuff" dimension, the political issues, has grown from being a whimpering child, barely audible over the din of more demanding technical problems, to become the shrieking, problematic adolescent very nearly drowning out all other voices. No longer can this problem child be pacified. No longer can the political dimensions of a project be viewed as irrelevant fluff.

If there is one point that this paper hopes to convey, it is that the dynamic of the development of MSW Incinerator Facilities has transformed from a black and white picture defined by technology issues into a blurry, hazy impression of greys which is governed by politics. Marshall McLuhan once said, "The medium is the message because it is the medium that shapes and controls the search and form of human associations and action". The medium is no longer numbers. It is now politics, environmental issues, hazardous wastes, landfill availability, tipping fees, trucking routes, recycling, and yes, somewhere at the bottom of the list, technology.

The message about the "soft" political issues is what will convince the community to allow the project to go through or not. A new approach has to be taken by parties attempting to develop

MSW incinerator facilities. It is no longer possible to "railroad" projects through the process. Although the problem of waste disposal continues to grow at an alarming rate, the "not in my back yard" philosophy has grown at an equally alarming rate, thus intensifying the need to deal with these communities in a sensitive and creative manner.

Developers and engineers have to take a more active role in working with the community. This will enhance their ability to tailor their designs to the particular community. Therefore, it is important to define the soft side of development as the process by which the relevant public is educated, informed and wooed so as to elicit a positive perception in their mind regarding a particular project. Does it sound a bit Machiavellian? Perhaps so, but in fairness to Niccolo Machiavelli, if he were operating today he would probably be a much sought after management consultant charging the princely sum of $300 an hour, or higher.

Public Relations then is the art of molding public perceptions to achieve the desired outcome. Those who deal with the public must always remember that to a remarkable degree perception is reality. Recent events on Wall Street bear eloquent testimony to the power of perceived reality to dramatically affect actual events.

It is also true that there are always two ways of looking at a matter. I am reminded of a story told about David of David and Goliath. It was observed that there was one big difference between David and the rest of the Israelite army. Whereas the others looked at Goliath and thought, "He is too big to hit," David looked at Goliath and thought, "He's too big to miss!" So as I begin to list some of the many groups that need to be dealt with, let's try to keep an optimistic outlook as David did.

WHO RECEIVES THE MESSAGE?

For starters let's look at the different groups whose support should be wooed in order to build an MSW incinerator facility. There are city councils or regional cooperatives that must be courted. Neighborhood organizations provide a good conduit for grass roots support. Local politicians are important contacts that must be stroked and whose support could be crucial to a project. Organizations within the region affected by the project need to be either recruited, neutralized or, at a minimum, defanged. It is important to work with interest groups early for experience has proven that early stage inclusion flatters its participants while last minute bandwagon loading greatly diminishes enthusiasm and sometimes is out right offensive. It is very important to build interest and support within the business community as well. Pro-growth forums, the chamber(s) of commerce and others are all fertile territory for planting the seed of development.

Of course, the more difficult foundation to lay down is the foundation that rests in the loose sand that is easily washed away by the anti-pollution, pro-environmental and no-growth groups. Their negative concerns need to be addressed at early stages in the process as well. Where the opposition is entrenched and belligerent it is vital that responsive steps be taken. One key step is to foster the development of a strong and credible support group from within the community. It is very important to avoid a situation in which the developer is drawn into a conflict situation with the local interest group. In such a contest the developer will appear like the carpet bagger, an exploitative outsider moving in avariciously on the homefolks. As you can imagine when the battle is viewed in these terms the developer almost always loses. Rather, local groups must be mobilized as an independent force arguing in favor of the project. Thus the issue is no longer one of the hometown vs. the outsiders but rather a conflict between different factions or interest groups within the community itself. This enables the developer to remain somewhat removed from the fray and on occasion even to appear as a positive force for compromise and conciliation. It is important to understand the opposition for only then can one anticipate the proper course to take in order to circumvent the obstacles that such groups will invariably throw in the project's path.

This is not to say that a negotiation should take place at the outset of contact with a community. Rather, it is important to give all groups the feeling that their views and concerns are vital to the development of the project without actually letting them dictate the program. It is important to remind oneself at all times that in the early stages the developer's role is that of diplomat. Remember that a diplomat has been defined as one who can "be disarming even when his country isn't." This kind of project requires a tremendous amount of diplomatic skill.

By now it should be obvious that the process being described is very similar to the steps a political candidate goes through just prior to announcing his or her candidacy. The waters are being tested to determine how well the community will receive the project.

This early grassroots work is critical from another perspective. It gives the developer and the leaders in the community the time to reconsider certain aspects of the facility's design or impact on the community. Questions of siting, noise, truck traffic, etc... all need to be tested and responses from the local groups need to be evaluated to determine the proper strategy once the decision has been made to develop the project.

It is time to invest in the ground work necessary to move the project through the meriad approvals from zoning to operating permits, from regulatory to environmental approvals after a favorable enough response from the community has been identified which gives confidence that the project has a reasonable chance of being built. Elected officials on the boards and commissions that will preside over the approval process are a critical group to contend with although one hopes not in a contentious manner. Interesting enough, those who serve in elected positions on boards are usually laypersons who have to rely on the paid staff for their information about the acceptability and desirability of different proposals that come before them. In dealing with the staff, there is more liberty and leeway. In general the developer can afford to be a bit more collegial and personal in dealing with staff than when he or she is dealing with the elected officials and I believe it advantageous to do so. It is advisable to make a courtesy call by way of introduction but there is no need to stop there. Solicit the staff's advice on concerns they feel needed to be addressed in the proposal. Not only does this make the staff feel important, which they are, but it also yields a wealth of valuable information on where the minefields up ahead might be located. In many respects the staff has a far better idea of what the developer is up against than he or she does. Conversely, any information, data, statistics, and materials that can be supplied to the staff to make their job easier usually stands the developer in good stead.

The media is an ever present force and when I say force, I mean FORCE. It stands in a unique position to influence public perceptions and this position derives from the fact that, rightly or wrongly, the media, as a relatively disinterested party, is viewed as an objective and therefore reliable source of information on the merits and demerits of a project. People don't trust politicians, they don't trust lobbying groups, they don't trust businessmen or developers but they generally trust their local newspaper. This can make the media an extraordinarily useful tool if one handles it properly -- but beware, this is easier said than done. The first reality that one must bear in mind is that all newspapermen (as well as other media representatives) are innate skeptics and even cynics. They dislike being patronized or handled but they are not above trying to "handle" you. The second truism to bear in mind is that eventually everything "will out". There is no way, in the current climate, to keep any aspect of a project shrouded or under wraps.

Given these assumptions, the best way to proceed in dealing with the media is to be candid, professional and forthcoming, within reason. This sort of approach tends to engender confidence and a sense of trustworthiness. If a reporter finds that he or she can depend on you to provide accurate and thorough information, they will be more inclined to rely on you as an information source and this can be of enormous significance. Remember that the difference between favorable and unfavorable reporting of your project may literally turn on a handful of descriptive words used in a news story and the way the headline is phrased. It makes all the difference in the world for a developer to be described as "candid and responsive" vs. "guarded and defensive" and in large measure the way the media perceive their own dealings with you will mold their perceptions of your dealing with the community in general. Be wary of disinformation and praise editors when good and fair articles are printed. Most importantly, when composing press releases or writing brochures to distribute to the community, talk in terms that are understandable and make sense to the public. The title of this conference is a good example of a confusing term. Municipal Waste Combustion is not nearly as widely understood as "Burning City Trash". In an era when the rule seems to be to euphanize every remedial act performed by a city's public works department it is helpful to remember that some euphanistic terms actually sound worse than the activity they are trying to describe. If the "term of art" is too vague or causes confusion it can easily create suspicion and doubt.

All of what has just been described may sound like an insurmountable mountain to climb at first but remember, it represents the fertile field or the barren plain upon which the project will be built. Without the grassroots support a project has a slim chance of ever breaking ground. With the proper nourishment, it can possibly grow into a successfully developed MSW incinerator facility. What follows are two brief case studies of projects that met very different outcomes, because of the differences in their "soft" sides.

TWO BRIEF CASE STUDIES

ROCKINGHAM COUNTY PLANNING COMMISSION

The State of New Hampshire, in 1983, revised its legislation concerning solid waste management by adopting a legislative statute annotated RSA 149-M. This legislation called for the formation of Regional Waste Management districts throughout the state that would each be responsible for handling the solid waste generated within their borders. The statute has been in existence now for well over five years and so far only four plans have been submitted to the state as required under the legislation. Of those four plans Exeter is the only plan that has received state approval.

There are many problems endemic in the approach taken by the State of New Hampshire. To begin with, New Hampshire has provided very little support on the state level for any action taken at the regional level. Both before and after the legislation was passed there was inadequate ground work laid on the local level. Suddenly twenty one communities on the sea coast were thrust together to try and forge a cooperative approach to disposing of their common waste. Economic and demographic differences among the towns caused tremendous rifts between the communities. Meanwhile, the statutes guiding the planning commissions were too weak to force non cooperative communities back in step with the group. Eventually, through the heroic efforts of a handful of individuals, a coalition of thirteen towns out of the original twenty one participants survived the melee . It is interesting to note that the rational of the eight towns that abandoned ship had to do either with their location on the periphery and their consequent decision to go outside the region to dispose of their waste or it had to do with their central location and consequent concern about avoiding the possibility of having the MSW incinerator facility located within their boundaries.

Throughout the entire process the state did not offer direction beyond the aforementioned "toothless" statutes. Furthermore, it was very difficult for any kind of consensus to be reached due to the lack of money for proper publication of information for the citizens. Too often the media misquoted or misrepresented the actual process due to biases inherent in the publication or by lack of understanding of the subject matter. Too little time was spent on education which caused tremendous communication problems.

During the heat of battle a private developer entered the scene and proceeded to, outside of any of the politically derived boundaries, option land and pursue permits for an incineration project. The total disregard for the regional plan and insensitivity to the citizens who had been investing their lives in the process generated a very understandable negative reaction. A reactionary group, S.O.S., (for Stamp Out Smokestacks), mobilized due to the incorrect perception that the only approach to disposing of MSW was by incineration. This polarized the region's participants and put the regional planning commission into a defensive strategy from which it never recovered.

RESOURCE RECOVERY CO-OPERATIVE, CONCORD, NH

Conversely, in another part of the state near its capitol, a decision had been made by the region's largest community to support the development of a waste to energy incinerator before it even went to referendum. Concord, the state's capitol, was so committed to seeing a regional refuse burning facility located within its borders that it forced a developer who intended to develop a wood chip burning power plant near the site of the incinerator and who was ready to apply for an air permit before the incinerator was, to pull its permit and locate outside of the town boundaries even though its electricity was to be sold to Concord. This strong sense of urgency and decisiveness was instrumental to the success of the Concord regional facility. More importantly, there was a sizable (for New Hampshire) community that was willing to fight ardently for the development of a facility within its borders.

Alas, it is the former example of the Rockingham Planning Commission and the Souteast Regional Solid Waste District that is the norm and not the latter success story. The developer, in both cases, did little to enhance his or her position within the community and in many instances created more trouble for the project than good will.

CONCLUSION

At a time when America is awakening to the realization that the environmental excesses of the past are coming home to roost, developers of projects that will deal with the growing sea of waste must grapple with a much larger perspective than they previously brought to their work. So much efficiency has been squandered in the past. So much neglect toward the health hazards to individuals, let alone this small planet, has been tolerated by the public and its elected officials. A new day has arrived where the developer who wants to incinerate municipal solid waste must be someone who is committed to a much broader picture.

Part of this broader picture includes an increased involvement in the legislative process at all levels of government -- but particularly at the national level. The business and scientific communities both have a vital role to play in helping formulate a serious national policy for dealing with what is obviously a serious national problem.

And, of course, at the heart of any long-term solution lies education. Re-educating generations of Americans raised on the throw-away philosophy of past years will be no small task but it can and must be done.

It is a daunting undertaking in this day and age to endeavor to develop an MSW project. Indeed, the gauntlet that must be run seems almost overwhelming but it is doable and I firmly believe that the value in knowing in advance what you are up against is what helps ensure that only the really good projects will be built and in the end that kind of selection process is good for business, good for communities and good for this precious land we share.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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DEVELOPING, PROMOTING AND ENDORSING INCINERATOR TECHNOLOGY

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ABSTRACT

Municipalities across Canada and the United States are forced to directly address the issue of incineration every time they carry out a municipal waste management planning strategy. Provincial and Federal authorities do not appear to have taken as decisive a political stance for or against incineration as a component of waste management planning.

With growing public sensitivities towards environmental issues and a looming garbage crisis, I believe the time has come for national consensus on provincially imposed standards and a strong, decisive political statement on the future of incineration technology.

The opinions in this paper are Ms. McGee's personal opinions and are not to be attributed to the Regional Municipality of Ottawa-Carleton.

As staff counsel to the Regional Municipality of Ottawa-Carleton's Waste Management master planning process, I am repeatedly left with the feeling that we are "reinventing the wheel" in waste management planning. Although only a handful of Ontario municipalities precede us in carrying out waste management planning, I am left with doubts about what is expected of municipalities and what the appropriate municipal responsibility should be.

The current model is this. We have municipal residential and commercial waste streams being disposed of largely at municipal facilities. The operation of those facilities is strictly governed by Provincial legislation. The planning and development of new facilities is also strictly controlled by provincially imposed legislative processes.

Unlike non-waste related municipal land use planning, the provincial environmental process does not begin with the presumption that municipal councils carry out proper planning in their waste management decisions. Therefore, at the end of a protracted, expensive selection of site and technology, the whole process begins anew before an administrative tribunal whose mandate is to begin with no preconceived notion that the municipality made the right choices (even if those same choices were made by a previous proponent).

Added to this dilemma is the public pressure the municipality faces to integrate 'experimental' technologies into their waste management master plan. While test runs and models may have provided encouraging results, municipal governments are usually forced to make the politically discomforting decision to choose the proven technology over the promising experiment. The reality behind this decision is both the uncertainty of receiving provincial approval for unproven technologies and the huge expense in running that risk.

Incorporation of experimental technologies inevitably also addresses funding. Municipal budgets do not have the dollars available to underwrite waste management experiments. More importantly, I don't believe it is the municipalities who should be expected to promote and subsidize research and development. That responsibility lies clearly at the Provincial (or State) level. The argument might even be made that research and development of new waste management technologies is also of national importance and therefore requires adequate federal incentives.

RESPONSIBILITY FOR SELECTION AND APPROVAL OF INCINERATOR TECHNOLOGY

(1) THE CANADIAN REGULATORY FRAMEWORK

Under the current Ontario system (with which I am most familiar) the division of responsibility is fairly clear. Municipalities identify a need, then usually proceed to develop and manage their own waste facilities. In many instances, the municipalities have a veto power over the establishment of new or expanded competing facilities.

At the provincial level, the government sets the legislative framework within which municipal waste management facilities must be developed, built and maintained. They provide ongoing supervision of facilities through an approval renewal process and active investigation and enforcement branches.

Federal jurisdiction in Canada is limited to transboundary matters of both an international and interprovincial nature. The Federal government also addresses product control with a view to its eventual disposal; public safety from the perspective of national health and environmental impacts; and of course, activities on Federal Crown Lands and Federal facilities.

The Federal Government has been most active in developing a unified definition of hazardous waste, plotting inventories of waste, funding research projects in waste management and monitoring and controlling the transboundary movement of waste. Traditionally, I believe the Federal Government saw its role as a provider of information and guidance to industries and provinces to promote the development of a uniform hazardous waste management system in Canada.

Across Canada, each province takes responsibility for the regulatory framework governing the planning and development of municipal waste management facilities. The truth is that the approval process and efficiency standards differ dramatically from one part of Canada to the other. As a general comment, I think it is fair to say that those provinces most often labelled as the "richer" provinces have the most sophisticated and onerous statutory requirements. In an era of heightened environmental sensitivities, it is inconceivable that some provincial governments have yet to legislate emission control guidelines and waste management standards.

In the province of Saskatchewan, permits for incinerators are issued by the Ministry of the Environment under the Air Pollution Control Act following submission of information by the proponent. There are no public hearing requirements.

The Province of Prince Edward Island has one operating incinerator (with no emission control devices) owned by the provincial government. The province has no legislated emission standards, although an individual was recently hired to create such standards. Preliminary estimates suggest that it will cost more to bring the plant into compliance with new emission standards than it did to build the plant in the early 1980's.

In the Province of Nova Scotia, the Ministry of the Environment carries out an internal review of applications for incinerators. The legislation does not include a public hearing. There are four operating incinerators in the province ranging from 15 to 200 tonnes per day. The provincial government intends to legislate emission control guidelines at least for incinerators with a capacity of more than 25 tonnes per day. While the legislation exists to permit the Minister of the Environment to establish emission standards, no legislated standards have been enacted. Standards are currently imposed as a matter of departmental policy through the internal application approval process.

The approval process in the province of Alberta is an application containing certain required information to the Department of the Environment. There is no public hearing. The Province's Clean Air Act requires a permit to construct the facility and a licence to operate. While there are provincial regulations with respect to visible emissions, the Department looks to other agencies (such as the Environmental Protection Agency) for the development and monitoring of appropriate standards.

Finally, in British Columbia the Ministry of the Environment regulates all incinerators except those in the Greater Vancouver Regional District (G.V.R.D.). However, the G.V.R.D. and Provincial certificates and standards are identical. Both bodies work closely with the Canadian Council of Resource and Environment Ministers (CCREM) in the establishment of emissions standards. An incinerator opened recently in Burnaby, B.C. to serve the G.V.R.D. is currently being tested. Another plant of comparable size is in its planning phase for Coquitlam, B.C.

What the current state of regulatory guidelines point to is a clear lack of unified standards and approval requirements. While Ontario imposes a protracted public hearing process, many other provinces have no such requirement. Emissions and performance standards may be unlegislated departmental guidelines or strictly spelled out provincial regulations.

I do not wish to embark on a detailed constitutional debate as to what level of government should be providing the regulatory structure for the planning, approval and operation of incineration facilities. I have no quarrel that the individual provincial authorities are the most appropriate body. I am surprised by the distinctions from province to province given both the place of the environment in public perception and the existence of interprovincial networks such as The Canadian Council of Resource and Environment Ministers.

Brief mention has been given to the role of the Federal Government in waste management and technology research. In terms of a significant political contribution to incineration, I believe the Federal Government is to be applauded for its work in the National Incinerator Testing and Evaluation Program (NITEP).

The 1986 NITEP program in Quebec City evaluated the capabilities of two different emission control systems to remove particulate, acid gases, heavy metals and trace organics. Three distinct sampling locations were selected to determine removal efficiencies.

In an earlier NITEP project carried out in Prince Edward Island, the focus was on establishing the technological capability of two-staged controlled air combustion incinerators (with fuel gas recovery) to handle municipal solid waste destruction and the emission levels resulting from such operations. As recently as 1988 testimony before an Ontario tribunal[1], NITEP results were held out as the most up to date and reliable data available when determining emission rates and identifying contaminants with the greatest impact on human health and the environment.

In making mention of the Federal NITEP program it is interesting to note the force behind its inception. There was growing focus in Canada (as elsewhere) on energy from waste technologies as a means of reducing the need for municipal landfills. Federal authorities in the early 1980's noted with some concern, the variation in reports on emission standards and disagreement on health and environmental impacts.

Through an Expert Advisory Committee on Dioxins, the Ministers of Health and Welfare Canada and Environment Canada recommended that the best available incineration technologies in Canada be identified and ranked through an environmental monitoring program. The committee further recommended that greater efforts be made to define optimal design and operating conditions for municipal and industrial incinerators to reduce or eliminate emissions of concern. In response to the Committee's report, NITEP was established by Environment Canada.

If I return now to my premise that provincial bodies should be left with the responsibility for regulation of incinerator technologies and that the current state of affairs is unsatisfactory, where then do we go from here?

Though municipal waste management appears to be a local problem falling within distinct provincial boundaries, municipal waste management solutions may have transboundary impacts requiring interprovincial consensus. Air emissions, leachate migration and impacts on the natural and social environment are not inclined to be constrained by provincial boundaries. The environment is now a national issue and activities impacting on our environment must have a national perspective even when they are a provincial concern.

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[1] Reasons for Decision and Decision by The Joint Board under The Consolidated Hearings Act, 1981 for "A Proposed Energy from Waste Facility, An Undertaking by SNC Inc." In the Regional Municipality of Peel, Ontario, October 24, 1988, at p. 38.

The Expert Advisory Committee on Dioxins had the right idea in recommending a national inventory of best available technologies and optimal design and operating conditions. They did not presume to impose those standards but rather create a national resource tool.

I believe we must work towards provincially imposed national standards for environmentally sensitive activities. There must be an inter-provincial consensus and pooling of resources to achieve solutions to waste management problems. Provincial regulatory frameworks must reflect both the national significance of waste management practices and the local characteristics of waste management needs.

FUNDING AND ENDORSEMENT OF INCINERATION TECHNOLOGIES

There is uniform agreement among Canadian provincial regulatory bodies that responsibility for controlling the planning and construction of incinerators must remain in their hands. However, when discussing the issue of funding research and development, only the province of Ontario has a program in place to encourage new work in the area of energy from waste. The Ministry of Energy, Mines and Resources for the province of British Columbia is currently developing a program to assist energy from waste projects.

In Ontario, the Ministry of Energy's program budgeted twenty-five million dollars for a five-year period to cover 15% of capital costs of an energy from waste project up to a maximum of six million dollars per project. Projects must involve proof in design, best available control technology, guaranteed energy sales and a minimum tipping fee of $30.00 per ton.

The major drawback to the program is that final commitments to assistance are not made until all necessary environmental assessment approvals have been obtained and all eligibility criteria, set by the Ministry of Energy, have been met. Notwithstanding the approval prerequisite, the Ministry of Energy recently provided $350,000.00 to Petro-Sun/SNC Inc. to assist with the preparation of material for their application to the Environmental Assessment Board.[1]

With the sole exception of Ontario (and British Columbia's planned program), no other provincial regulatory authorities have funding programs in place for energy from waste research and development. The general feeling seems to be that funding and grant programs are costly and should be left to 'someone else'.

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[1] Op. Cit. "A Proposed Energy From Waste Facility, An Undertaking by SNC Inc."

As indicated, provincial governments in large measure, appear to have no special private-sector funding available for research into new technologies, and emission standards, capital projects and 'experimental' disposal techniques. In addition, with the exception of provincial involvement in the federal NITEP Program[1], there appear to be no provincially driven 'in-house' development projects and construction. Given the current crisis state of much of our municipal waste management, it is inconceivable to me that there are so few dollars available for EFW research and development.

Even in Ontario, one cannot help but be struck by the 'mixed messages' being sent by different ministries within the Provincial Government. While the Ministry of Energy has identified twenty-five million dollars for energy from waste capital project assistance, the Minister of the Environment has repeatedly suggested a cautious approach in incorporating incineration technologies into municipal waste management master planning. In a statement from the Ontario Minister of the Environment on March 18, 1987, he said:

> 'Energy from waste facilities represent an alternative to both landfill and conventional sources of energy, but concerns about health and environmental effects require comprehensive evaluation prior to environmental approval.'

If the Ontario program is the most positive incentive available in Canada for the development of energy from waste technology, then their guarded support provides little encouragement to research and development or to potential EFW proponents. A proposed energy from waste facility approved by The Joint Board in October, 1988[2] persevered through three to four years of planning, seventy-seven days of hearings and the bankruptcy of one of the private proponents (Petro-Sun International).

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[1] The National Incinerator Testing and Evaluation Program: Air Pollution Control Technology (NITEP). A joint program of Environment Canada with Flakt Canada Ltd., conducting a series of comprehensive tests on two pilot-scale air pollution control systems at the Quebec City Municipal Solid Waste Incinerator Plant.
The National Incinerator Testing and Evaluation Program: Two-Stage Combustion, a project of Environment Canada conducted at the provincially owned municipal solid waste incinerator in Parkdale, Prince Edward Island.

[2] Op. Cit. A Proposed Energy from Waste Facility, An Undertaking by SNC Inc.

While the regulatory frameworks are in place across Canada for the approval of new incinerators, there are in fact very few being built. Though most provincial authorities agree that landfill is not a "solution" to our current garbage crisis, it is in fact largely landfills that municipalities are steering through provincial environmental assessment approvals.

Municipalities, I believe, are under both financial restrictions and disposal capacity deadlines to replace their current waste management infrastructures. Without consensus from the regulatory authorities to encourage new construction of energy from waste facilities, then most municipal proponents don't believe it is there responsibility to underwrite new developments in energy from waste technology. The approval process in Ontario is very lengthy and the future of provincial support uncertain.

It is little wonder that the commitment to EFW research and development does not appear to be strongly encouraged at the municipal level at this time. I don't in fact believe that funding should be generated at the municipal level. If the provincial authorities claim responsibility for the regulatory framework within which EFW development takes place and impose a regulatory framework that allows for approvals, then it is that same regulatory body that should be encouraging (or discouraging) research in better technologies.

I believe that the time has come for politicians to clearly state they do or don't support both the current energy from waste technologies and the initiatives underway to refine these projects. It certainly is not a palatable prospect from the perspective of the politician or government pressured to that point. However, in the absence of strong political encouragement to continue EFW research and development, I don't see why municipalities should run the risk of promoting a new EFW facility in a vacuum.

Making a strong political statement in support of incineration is somewhat akin to a definitive public policy on abortion. You can guarantee that your decision will strongly divide the electorate into two opposing, emotional camps. Therefore, the easier route appears to be a cautious middle line where you parade the 'common' concerns surrounding incineration (toxic ash, emission contents) and also express cautious optimism for the future of incineration technologies in waste management.

This current state of affairs leaves it up to municipal proponents to make the political decisions to decisively support or reject incineration in the course of their waste management planning. It therefore becomes the municipalities that take the strong stands for or against incineration by deciding whether or not to include it as a component of their waste management solution.

If a public (or private) proponent decides to 'bite the bullet' and proceed through environmental assessment approvals, what then is and should be in store for them? Take by way of example, the most recent decision in Ontario[1]. The proponent, SNC Inc., proposed the construction of an EFW facility to incinerate 364 tonnes per day of 'acceptable waste' defined as, 'all solid waste, garbage, trash, rubbish and refuse which is normally collected in accordance with municipal by-laws ....'[2] The facility was to include 'fuel enhancement equipment' designed to remove approximately 40 tonnes per day of metals, glass and grit from the waste before incineration.

Incineration was to take place in four modular, two-stage, controlled-air combustion incinerators each with a rated capacity of 91 tonnes per day. There would be two sets of air pollution control equipment. Each system was to comprise a gas conditioning tower, a fabric filter baghouse and an induced draft fan.

There were a number of known political standards that the project was expected to meet. The Ministry of the Environment's (MOE) policy, 'The Combustion of Refuse in Incinerators' establishes design and operating guidelines for incinerators which burn domestic, commercial or non-hazardous waste. The policy provides detailed minimum design parameters for incineration temperature, residence time, oxygen availability as well as a number of operating, control and monitoring conditions within which the parameters are to be met. Ministry experts testified that the proposed facility was capable of meeting this policy.

On the issue of air emissions, the hearing Board reviewed Ontario Regulation 308 (the principal regulatory instrument regarding air quality) and the government's 'List of Ambient Air Quality Criteria, Standards, Tentative Design Standards, Guidelines and Provisional Guidelines'. Consensus at the hearing was that the proposed facility could also meet these legislated standards.

MOE experts also introduced a departmental policy requiring the installation of state-of-the-art air pollution control systems on all new incinerators and an 'Interim Guideline' proposing maximum particulate emissions and minimum HCl removal for all new air pollution control systems.

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[1] Op. Cit. 'A Proposed Energy From Waste Facility, An Undertaking by SNC Inc.'

[2] Ibid at page 6.

These proposed guidelines were the result of a government review of the capabilities of state-of-the-art technology. It is interesting to note that the standards imposed on SNC Inc. as a condition of approval were not those found in the air quality regulations but rather those presented in a MOE Discussion Paper containing 'revised models' and 'draft amendments'[1] to Regulation 308. Furtunately, the proposed technology was state-of-the-art and therefore was capable of not just meeting but exceeding the current legislated standards.

Whether or not the SNC Inc. energy from waste facility will now be built, remains to be seen. In the framework of Ontario environmental assessment approvals, the final issuance of certificates to proceed remains in the hands of the Minister. As long as the regulatory framework is such that technological capacity can exceed provincial guidelines then there will be the occasional proponent who will persevere through the planning and approval process. Each time this happens, the Provincial government will find the issue of incineration and energy from waste right back in their lap. To date, these proponents have faced so many regulatory hurdles and so little encouragement to proceed, it is no wonder that a province the size of Ontario has reviewed only two new incinerator proposals in the past six years.

I hope in describing the SNC Inc. protracted approval process, I have come full circle to expressing my frustration with the current promotion and approval processes for incineration technologies. In the face of growing environmental sensitivities, where is the decisive political leadership to guide us into twenty-first century waste management options? Whether the answer is landfill, recycling, composting, incineration or something else there must be a political commitment in terms of dollars and policies to encourage research and develop workable solutions. Walking a politically palatable 'middle line' between support and rejection of incinerator technologies serves no positive ends.

I don't believe it is the responsibility of municipal proponents to 'reinvent the wheel' every time a waste management option is carried through environmental approvals. Unfortunately, provincial authorities will not acknowledge the 'acceptability' of any current incinerator technology nor do they appear willing to adequately fund the research necessary to compile convincing data for specific technologies. Regulators talk in terms of standards rather than methods of achieving those standards. There are no provincial policy statements endorsing any specific incineration technology or air emission control system. Everything is minimum and maximum operating standards.

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[1] Op. Cit. 'A Proposed Energy From Waste Facility: An Undertaking by SNC Inc.', at pp 64-65.

Nowhere in Canada can I find a provincial authority with a regulation or policy that says: "We think this technology works. We know this system meets and exceeds our standards. We encourage those proponents considering incineration to incorporate this technology into their waste management planning. Come to us with a proposal for this technology on a suitable site and we will give our blessing."

I can't even find a provincial environment Ministry that is willing to announce: "We think this technology works. We will be a partner in research do prove this technology can meet and exceed our standards. We will support a proponent who shows us the results we believe this technology is capable of obtaining. Incineration is a viable component in our future waste management needs."

I would not even quarrel with a regulatory authority that was willing to say, "As of April 1, 1989 we endorse this technology as an environmentally acceptable means of incinerating municipal waste. In light of our commitment to continually reduce and eliminate the discharge of toxins to our environment, our current support of this technology will not hinder our future imposition of more severe design standards reflecting the evolution in and commitment to increasingly sophisticated technologies". Unfortunately, I don't yet see even that qualified commitment to a particular incineration package.

Therefore, if I were to represent a client proposing an incineration facility identical to the 1988 package presented by SNC Inc. to the Joint Board under Ontario's Consolidated Hearings Act, I would be put through the exact same approval and hearing process. There would be no concessions made for the decision of the Joint Board, the testimony of acknowledged experts or the position of the Ministry of the Environment with regard to the SNC Inc. proposal.

In my mind, this is simply not a logical use of public funds. To put this second hypothetical proposal through a repeat performance of an identical predecessor points to one of the flaws in the current positions taken by regulatory authorities. If municipal authorities could look to political endorsements of certain generic technologies, then we would save millions of dollars on repeating the research of benefits, disadvantages and performance standards of the alternative methods of incineration every time a proponent seeks approval for a facility. Since we get no decisive political guidance, each municipal proponent is forced to review the complex issue of incineration as though no one before them has ever carried a proposal through environmental assessment approval.

In Ontario, the provincial government provides partial funding for a municipality's waste management, environmental planning process. The provincial treasury funds the administrative hearing Board that reviews these municipal applications. In addition, an extensive Environment Ministry provides planning advice, reviews environmental assessment documents and participates throughout the lengthy administrative hearing process. I believe that some of those tax dollars could be better spent if the provincial regulators made a political commitment to particular technologies and then provided funding incentives to encourage development of even better versions of those facilities. As long as our regulatory authorities avoid an outright rejection of incineration, I believe they have a political and public obligation to overtly assist the development of increasingly sophisticated technologies and go on record with a decisive public position.

This same guarded approach to incineration is reflected in a public policy study prepared for the new President of the United States[1]. The report notes the shift in support away from garbage incineration. The attractiveness of initial projects was augmented by federally subsidized tax-exempt public financing and investment tax credits to private-industry proponents. More recently, tax reform legislation has restricted federal subsidies and the Report notes that concern is growing over incinerator air emissions and ash residues. Therefore, the policy study cautions that further research is needed to resolve technical problems and standards for air emissions and ash disposal.

The irony in the current situation in both Canada and the United States is the political conclusion that further research is needed while financial subsidies and incentives appear to be dwindling.

The public policy Report to President Bush endorses recycling and concludes,

> "it is absolutely essential that recycling be considered on an equal basis with other alternatives .... Data on existing recycling programs should be collected, and cost analysis of these programs performed and disseminated, presumably through research and education by EPA and relevant state agencies."[2]

Other than the suggestion that incineration requires further research, the Report does not provide recommendations for who should fund this or how it should be carried out.

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[1] "Project 88 Harnessing Market Forces to Protect Our Environment: Initiatives for The New President". A Public Policy Study sponsored by Senator Timothy E. Wirth, Colorado and Senator John Heinz, Pennsylvania. Washington, D.C. December, 1988.

[2] Op. Cit. "Project 88 Harnessing Market Forces To Protect Our Environment" at p. 71.

According to an article appearing in the Wall Street Journal[1], while the garbage crisis has encouraged the construction of seventy-three energy from waste plants across the U.S., more than three billion dollars in projects have been cancelled since the beginning of 1987.

CONCLUDING REMARKS

The message is becoming clearer. The enthusiasm for incineration in the 1970's as the best replacement for exhausted municipal landfills has waned. The current political consensus advocates the need for further research into such specific areas as air emissions and ash residues. No political consensus and few financial incentives exist to reinforce the recommendations for further research. Public policy and political consensus does exist in support of recycling initiatives and education. No political statement has been made specifically rejecting incineration as a viable option for future municipal waste management. While technologies exist to meet current performance standards, the political will to site a facility is generally absent when faced with strong public opposition.

Municipal governments carrying out waste management planning strategies are facing frustrated voters who want to see more innovation in waste management solutions. Municipal budgets can't afford to subsidize promising experimental technologies. To proceed through waste management planning with an incineration component encourages strong public reaction. There is no clear support or encouragement at the provincial and federal levels to incorporate incineration components in municipal waste management strategies. The public often believes that past incineration `mistakes' will be repeated in current technologies. As long as provincial and federal regulators fail to develop the policy and financial incentives that are necessary to promote research and testing, then municipal proponents are without the tools necessary to respond effectively to their opposition.

Municipal proponents have been left the task of proving and packaging incineration technologies every time they make a local political commitment to incineration as a component of their waste management solution. Now, I believe, the time has come for a national consensus on the role of incineration in our future waste management plans. Taking a firm political stand and supporting that decision with dollars and policies is essential to determining the role of incineration in our long term waste planning.

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[1] "Energy From Garbage Loses Some of Promise as Wave of the Future" Wall Street Journal, June 16, 1988, p. 1.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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ACID GAS EMISSIONS: RESULTS OF A SPRAY DRY SCRUBBING PILOT PLANT STUDY

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ABSTRACT

Burning of municipal solid waste (MSW) creates a wide range of particulate and gaseous emissions. Spray dry scrubbing, a technology employing gas absorption in a spray dryer contactor, followed by particulate removal in baghouse or electrostatic precipitator, is among the most efficient and cost effective way of meeting today's emissions regulations. To further provide information on the performance of dry scrubbing systems when applied on MSW combustion, a pilot plant test program has been conducted. The test facility is comprised of a 4500 acfm spray dry scrubber employing a single rotary atomizer and a pulse-jet baghouse for particulate collection.

The dry scrubbing system performance was studied as a function of important operating parameters. Overall, the spray dryer outlet temperature was found to have the greatest influence on HCl and SO_2 removal and/or lime consumption. Safe system operation from both a corrosion and material handlings standpoint can be maintained at spray dryer outlet temperatures down to 230 °F (110 °C). Furthermore, recycle of the baghouse products demonstrated a great potential for reduction in lime requirements, again with no indication of material handling problems. Finally, characterization studies were performed on the baghouse product using the EPA toxicity test procedure.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

INTRODUCTION

Disposal of municipal solid waste (MSW) is rapidly becoming a problem as existing landfills reach their capacities and establishment of new landfills is becoming increasingly more difficult. Incineration of MSW has gained support over recent years as a viable and safe way to achieve significant volume reduction and extend the life of existing landfills. Typically, incineration reduces the waste volume by 80 to 90 percent, extending the useful life of a landfill by 5 to 10 times. At the same time, incineration allows for significant energy recovery through production of electricity or district heating.

A number of pollutants are produced by incineration of MSW, ranging from acid gases (HF, HCl and SO_2) to heavy metals and organic compounds (dioxins, furans, etc.) The combustion process is important in controlling emissions of many of the organic compounds. However, good combustion practices are not a substitute for post combustion pollution control technologies and State and local regulations in the U.S. require the control of emissions from new municipal waste combustion (MWC) facilities. Dry scrubbing systems comprised of a spray dry absorber followed by particulate removal in a baghouse or electrostatic precipitator are commonly regarded as the best available technology for control of emissions from MWC facilities.

Combustion Engineering has long been active in the development of dry scrubbing systems for both coal and MSW fired applications. This paper presents the Combustion Engineering dry scrubbing process/design philosophy and the results from a dry scrubbing pilot plant study. The pilot study was performed using a mobile dry scrubber/baghouse system rated at 4500 acfm. The unit operated on synthetic flue gas obtained by burning natural gas in an incinerator and then mixing the flue gas with acid gases from cylinders and MSW fly ash from a local municipal incinerator. The permit for use of fly ash was issued by US EPA/ADEM under authority of the Resource Conservation and Recovery Act. Results include the influence of important operating parameters on the removal of acid gases and an investigation of the characteristics of the baghouse end product.

EXPERIMENTAL

GENERAL

The spray dry scrubbing process for control of acid emissions from MWC facilities is comprised of a reagent preparation system, a spray dry absorber followed by particulate removal in a baghouse or an electrostatic precipitator and an ash handling system. Typically, there is one spray dryer and one dust collector per incinerator train, a common reagent preparation system, and an ash transport system per multiple incinerator trains. The reagent is typically slaked lime. Flue gas from the incinerator at air preheater outlet temperatures is mixed with a fine spray of lime slurry in a spray dry absorber. A rotary atomizer has been proven effective and reliable for atomization of the slurry. The flue gas is adiabatically humidified and cooled to within some approach of the adiabatic saturation temperature by the water evaporating from the droplets. As the water evaporates, acid gases absorb and react with lime in the droplets. Typically, the slurry droplets are dried to less than a few percent of free moisture before leaving the spray dryer and are transported with the flue gas to the particulate collector device. Additional acid gas removal takes place in the dust collector.

The spray drying or cooling step is advantageous to the efficiency and economy of spray dry scrubbing systems in several ways:

- Removal of heavy metals and some organic substances are favored by a low flue gas temperature.
- The flue gas volume is decreased by the cooling step, resulting in reduced equipment size and cost. This is particularly important when an electrostatic precipitator is used for dust collection.
- The additional moisture from the cooling enhances removal of acid gases in the particulate collection device resulting in reduced lime consumption.

The Combustion Engineering spray dry absorber design features a bottom gas drawoff without any ash removal as shown in Figure 1. This design eliminates the potential problem posed by a spray dryer ash removal system. With proper

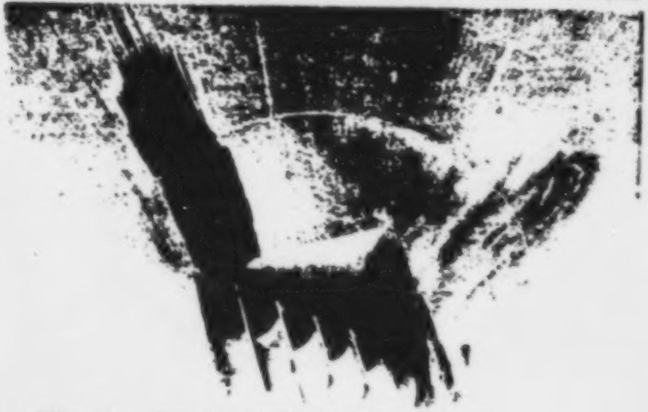

Figure 1. Spray dry absorber outlet at Valmy.

atomization and control of the spray cloud shape, absorber deposits are eliminated. However, should they occur during periods of extreme upset conditions, the normal corrective action is to restore routine operation and allow any deposits to dry and drop off into the outlet vanes. The gas velocity in the outlet vane area is high enough to erode and reentrain dislodged wall deposits. Of all the lime-based spray dry absorbers built according to this design, there have been no incidents where a dryer has had to be taken out of service for cleaning in order to restore normal operation (1).

A major design parameter reflecting the lime consumption required to achieve a specific removal efficiency is the stoichiometric ratio. As several gaseous compounds are removed by one sorbent in MSW applications, defining the stoichiometric ratio is not straight forward. The overall stoichiometric ratio (SR_o) is defined as:

$$SR_o = \frac{\text{eqv. CaO}}{\text{eqv. } SO_2 + \text{eqv. HCl} + \text{eqv. HF}} \qquad (1)$$

Although a number of chemical reactions occur simultaneously, the difference in reaction rates makes it possible to look on these reactions as consecutive. The most preferential reaction, which tends to occur first, is the removal of hydrofluoric acid (HF) which produces calcium fluoride. The next most preferential reaction occurs with hydrochloric acid (HCl). This reaction produces calcium chloride. The final reaction involves sulfur dioxide (SO_2) which produces calcium sulfite and to a lesser extent calcium sulfate. Because these reactions can be looked upon as pseudo-consecutive and the fact that both HF and HCl are removed close to 100 percent and independent of the removal of SO_2, the stoichiometric ratio can be redefined to reflect the reduced stoichiometric ratio (SR_r):

$$SR_r = \frac{\text{eqv. CaO} - \text{eqv. HF} - \text{eqv. HCl}}{\text{eqv. } SO_2} \qquad (2)$$

The reduced stoichiometric ratio reflects the ratio of lime to sulfur dioxide when the lime needed for complete removal of hydrofluoric acid and hydrochloric acid has been deducted. The relationship between the overall and the reduced stoichiometric ratio can be expressed as:

$$SR_r = SR_o * \left(\frac{\alpha}{2} + 1\right) - \frac{\alpha}{2} \qquad (3)$$

where, α represents the HCl to SO_2 concentration ratio.

Figure 2 shows the relationship between the overall stoichiometric ratio (SR_o) and the reduced stoichiometric ratio (SR_r) for a range of HCl to SO_2 concentration ratios (α). As the emission rate of SO_2 from the incinerator in most cases is minor compared to the emission rate of HCl, the overall stoichiometric ratio tends to mask the lime consumed for removal of SO_2. For example, at a $\alpha = 3.0$ (e.g. 600 ppm HCl and 200 ppm SO_2) and an overall stoichiometric ratio of 1.5, the reduced stoichiometric ratio would be 2.3 indicating a fairly large excess of lime not apparent from the overall stoichiometric number. The reduced stoichiometric ratio is used exclusively in this paper.

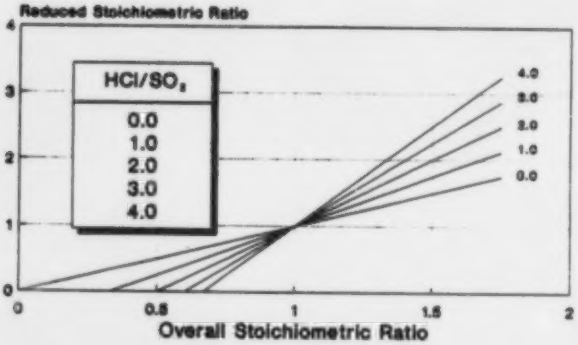

Figure 2. Relationship between overall and reduced stoichiometric ratio.

PILOT PLANT

A schematic diagram of the pilot plant used in this study is shown in Figure 3 and a detailed data sheet is given in Table 1. The pilot plant was comprised of a sorbent preparation system, a spray dry absorber followed by a baghouse for particulate removal, and a process control system. Simulated flue gas was produced by burning natural gas in an incinerator and adding acid gases (HCl and SO_2) from cylinders using calibrated flow devices. The humidity of the flue gas was controlled by injecting water into the incinerator. MSW fly ash from a local commercial incinerator was added to the flue gas shortly downstream of the incinerator. Successful redispersion of fly ash demands a flue gas free from entrained droplets. The spray dry absorber, rated at 4500 acfm, utilizes a single rotary atomizer for atomization and features the Combustion Engineering

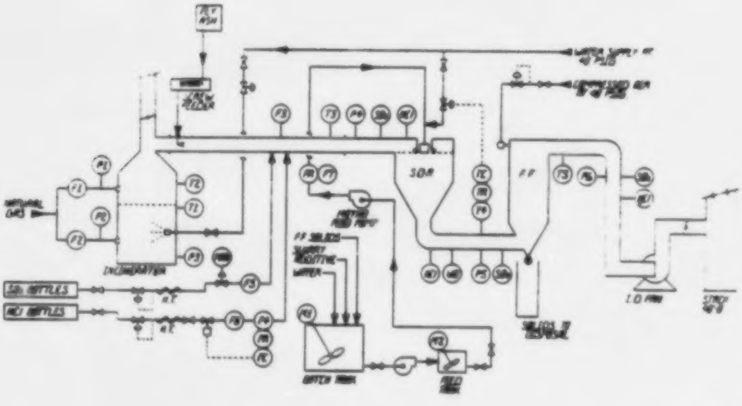

Figure 3. Schematic of the dry scrubbing pilot plant.

outlet design with no ash removal. The baghouse was of pulse-jet type with 64 six-inch diameter, ten feet long bags giving an air-to-cloth ratio of about 3.5 ft/min. A paste slaker operating at about 180 °F and 23 percent solids produced slaked lime with a maintained surface area of above 20 m^2/g. The desired stoichiometric ratio was obtained by controlling the slurry feed rate using a calibrated volumetric flow meter while the spray dryer outlet temperature was controlled with trim water. The slurry and the trim water feed streams were mixed shortly ahead of the spray machine.

ANALYSIS

This study focused on the performance of both the spray dryer and the overall system (spray dryer/baghouse). Hence, gas samples were continuously withdrawn from ahead of the spray dryer, at the spray dryer outlet, and after the baghouse. A standard continuous SO_2 instrument was used to monitor the SO_2 concentrations at all three locations. Hydrochloric acid was analyzed using a modified Method 6 train consisting of two impingers in series, both of which were kept in an ice bath. A chloride specific electrode was used to determine the chloride concentration in the impinger solutions. Originally, two impingers were used to assure complete absorption of HCl. However, it was found that one impinger was adequate as only an insignificant amount of HCl escapes the first impinger.

TABLE 1. DATA SHEET FOR THE SPRAY DRY SCRUBBING PILOT PLANT

Atomizer

Rotary atomizer; variable speed, direct drive
Wheel diameter; 6 inches
Wheel speed; 18,000 rpm
Tip speed; 471 ft/sec
Slurry flow rate; 0.2 - 1.6 gpm

Spray Dry Absorber

Diameter; 7 ft
Height; 9 ft
Volume; 224 ft^3

Fabric Filter

Bag diameter; 6 inches
Bag height; 10 ft
Total filter area; 1005 ft^2
Cleaning method; pulse jet
Cleaning pressure; 40 psi
Bag material; 16 oz Teflon® coated fiberglass

Experimental Conditions

Gas flow rate; 4500 acfm
Spray dryer residence time; 3.5 sec
Air-to-cloth ratio; 3.1 - 4.0 ft/min

ACID GAS REMOVAL

BACKGROUND

The importance of operating close to the adiabatic saturation temperature is well known from coal fired applications of the dry scrubbing process. A close approach provides a longer evaporation time in the spray dryer and hence a longer reaction time resulting in a higher sorbent utilization. Furthermore, a close approach enhances the removal of SO_2 in the particulate collector. Although this is also true for dry scrubbing systems applied on MWC facilities, the psychometric characteristics of flue gas from MWC facilities are significantly affected by the presence of HCl and the subsequent formation of calcium chloride.

Calcium chloride is a hygroscopic and a deliquescent compound which means that it strongly attracts water. Furthermore, the presence of calcium chloride in the droplets during the drying phase decreases the water vapor pressure and increases the wet bulb temperature of the droplets. Figure 4 is a psychometric chart for the air-water system on which the vapor pressure for a range of calcium chloride solutions has been plotted (2). The dot in Figure 4 represents a flue gas from a MWC facility with a temperature of 400 °F and an adiabatic saturation temperature of 145 °F, not taking the effect of calcium chloride into account. During drying of the droplets in the spray dryer, the concentration of calcium

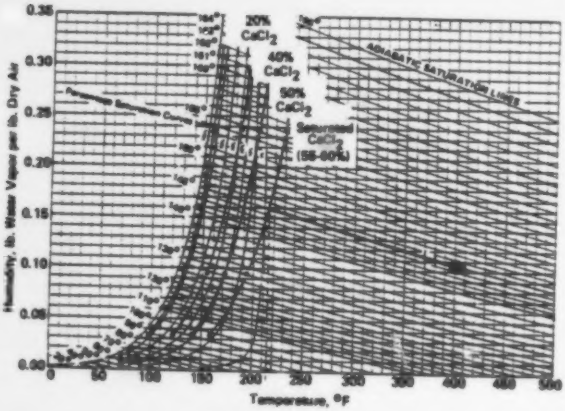

Figure 4. Effect of dissolved calcium chloride on water vapor pressure.

chloride will gradually increase until it reaches saturation. If the adiabatic saturation line in Figure 4 is followed to the curve for a saturated calcium chloride solution, a saturation temperature of about 210 to 215 °F is obtained. In other words, when a saturated solution of calcium chloride is reached in the droplets, the droplets will be at a wet bulb temperature of 210 to 215 °F and the flue gas temperature must be above this level for evaporation to continue. This agrees well with Combustion Engineering's experience with on-line commercial units which can and do operate down to 230 °F without any material handling problems.

RESULTS

A total of 79 test runs were performed utilizing the test conditions shown in Table 2. The introductory test runs focused on the effect of the baghouse pressure drop (dust cake thickness) on the removal of HCl and SO_2. For this purpose, the pilot plant was run at steady state conditions over a prolonged period of time with the baghouse pressure drop varying from 2.4 to 5.5 inches of water column. Typical results from these tests are shown in Figure 5. It was found that neither the removal of HCl nor the removal of SO_2 was affected by the cake thickness (pressure drop). This finding supports the theory that it is the outer layer of the dust cake facing the flue gas that is active in removing HCl and SO_2 while the remaining part of the dust cake is utilized to a point where it no longer contributes to further removal.

TABLE 2. EXPERIMENTAL CONDITIONS.[1]

	Min	Typ.	Max
SDA Inlet Temperature, °F	350	(425)	500
SDA Outlet Temperature, °F	240	(280)	320
Reduced Stoichiometric Ratio	0.5	(1.0)	2.0
Concentration of HCl, ppmv	300	(600)	900
Concentration of SO_2, ppmv	100	(200)	200
Fly Ash Loading, grains/acf	0.0	(0.0)	2.0

Another concern was the effect, if any, by the presence of MSW fly ash in the flue gas on the removal of HCl and SO_2. Again, the pilot plant operated at steady state conditions over prolonged periods of time with the ash grain loading as the only variable. Typical results from these tests are shown in Figure 6. It was determined that the presence of ash in the flue gas does not contribute to the removal of either HCl or SO_2 for fly ash loadings up to 2 grains/acf. The fly ash used in this study was acidic which is typical for MSW fly ashes and thus did not have any capacity for SO_2 removal. No ash was used in subsequent tests.

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[1] Conditions indicated with a bold print are those conditions commonly used. When other conditions are cited, this will be indicated in the text or in the figure captions.

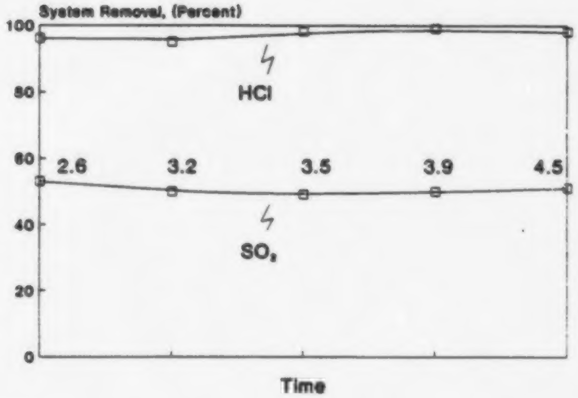

Figure 5. Influence of baghouse pressure drop on removal of HCl and SO_2. The numbers adjacent to the data points indicate the baghouse pressure drop.

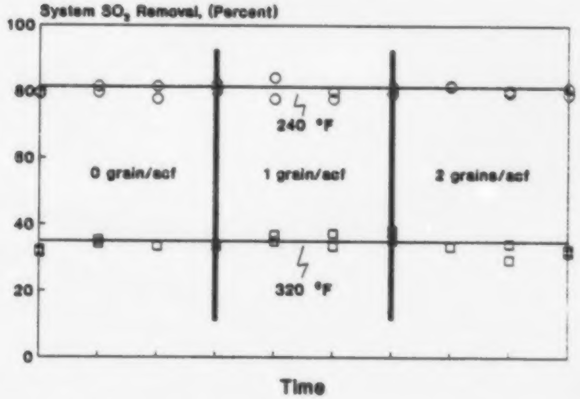

Figure 6. Effect of flue gas fly ash loading on SO_2 removal.

One of the most important operating parameters determining the lime consumption required to obtain a specified system SO_2 removal efficiency is the spray dryer outlet temperature or the approach temperature to the calcium chloride wet bulb temperature. A close approach temperature provides a longer droplet drying time and hence, a longer reaction time. Furthermore, the formation of calcium chloride in the droplets reduces the water vapor pressure of the droplets, resulting in an increased drying time compared to coal-fired applications. Shown in Figure 7 is the system SO_2 removal efficiency as a function of the spray dryer outlet temperature for a range of stoichiometric ratios. The strong influence of the spray dryer outlet temperature on the SO_2 removal efficiency is apperent. The pilot plant has been operated down to 220 °F without any indications of material handling problems. This agrees well with Combustion Engineerings experience from on-line commercial units that can and do operate down to 230 °F. Several key design features are required for successful operation at low approach temperatures, the most important being reliable and efficient atomization and a close control of the spray cloud shape. A rotary atomizer provides the uniform droplet size distribution and control needed for reliable operation, eliminating concerns regarding wall deposits and pluggage of the spray dryer outlet. Furthermore, any internal obstructions in the spray dryer are undesirable.

A second important parameter of the performance of dry scrubbing systems is the stoichiometric ratio, as shown in Figure 8. It is well known that an increase in the lime feed rate yields a higher SO_2 removal efficiency. This effect is particularly noticeable at lower stoichiometric ratios while at higher levels the effect diminishes and the lime utilization drops significantly. Figure 8 also shows the strong influence of the spray dryer outlet temperature on the SO_2 removal efficiency. For example, at a spray dryer outlet temperature of 240 °F, a stoichiometric ratio of 0.8 is required to achieve 70 percent removal while at a spray dryer outlet temperature of 320 °F, a stoichiometric ratio of 1.6 is required. For a spray dryer outlet temperature of 240 °F and an SO_2 removal efficiency of 90 percent, the corresponding stoichiometric ratio is about 1.4 while at a spray dryer outlet temperature of 320 °F, 90 percent SO_2 removal is not achievable at any lime feed rates.

Not shown in Figures 7 and 8 is the SO_2 removal efficiency across the spray dryer compared to the removal across the baghouse. Generally, the removal efficiencies across the spray dryer and the baghouse were comparable, based on their respective inlet SO_2 concentrations. For example, at a system removal efficiency of 70 percent, the removal efficiency across the spray dryer and the baghouse would both be close to 45 percent, although the removal across the baghouse would tend to be slightly higher.

The spray dryer inlet temperature was found to have a weaker influence on the removal of SO_2. The effect of the spray dryer inlet temperature was studied for temperatures ranging from 350 to 500 °F. A higher inlet temperature generally yields a higher SO_2 removal efficiency due to a longer droplet drying time. For example, at a spray dryer outlet temperature of 280 °F and a required SO_2 removal efficiency of 70 percent, a 350 °F inlet temperature would call for a stoichiometric ratio of 1.6, a 425 °F inlet temperature would require a stoichiometric ratio of 1.4, while at 500 °F the corresponding stoichiometric ratio is 1.1.

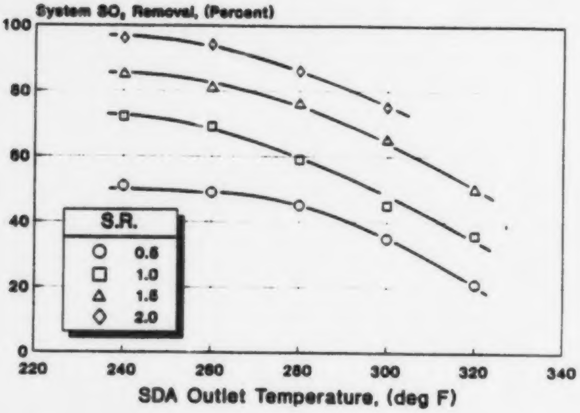

Figure 7. System SO_2 removal as a function of the spray dryer outlet temperature.

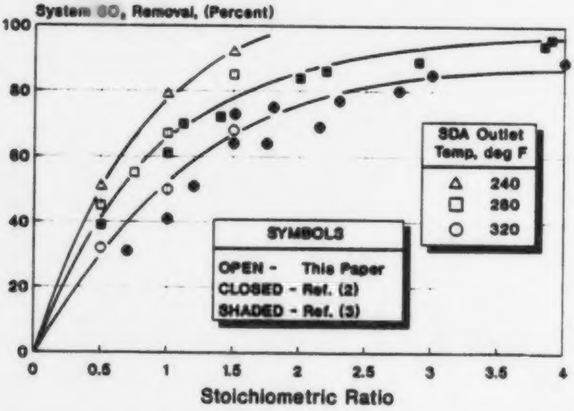

Figure 8. System SO_2 removal as a function of the stoichiometric ratio. Spray dryer inlet temperature 500 °F.

The concentration of acid gases in the flue gas emitted from an MWC facility is known to fluctuate substantially, mainly due to the inhomogeneity of the fuel. A dry scrubbing system employing a baghouse for particulate removal is uniquely suited to cope with fluctuations in the flue gas composition as the dustcake on the bags provides a substantial buffering capacity and can dampen the effects of rapid changes in the inlet HCl and SO_2 concentrations. Furthermore, by using the SO_2 stack concentration as a control signal for the lime feed rate, removal levels of both HCl and SO_2 can be maintained reliably due to the preferential removal of HCl.

The effect of the inlet concentration of HCl and SO_2 on the system SO_2 removal efficiency at a given stoichiometric ratio and spray dryer outlet temperature is shown in Figure 9. Four HCl concentration levels and two SO_2 concentration levels were tested; open symbols correspond to 200 ppmv of SO_2 while closed symbols correspond to 100 ppmv of SO_2. No effect on the system SO_2 removal efficiency was found for any of the combinations tested.

On coal-fired applications, it has been shown (4) that the effect of the inlet SO_2 concentration at a given stoichiometric ratio and spray dryer outlet temperature on SO_2 removal is merely a result of altered drying conditions. At a constant stoichiometric ratio, a higher inlet SO_2 concentration would require a higher lime flow , a slurry with higher percent solids, resulting in a shorter drying time and hence, a lower SO_2 removal. The HCl and SO_2 concentration ranges covered by this study resulted in only a marginal variation of the percent solids and hence no significant effect should be expected. Furthermore, the test results indicate that at even fairly low HCl to SO_2 concentration ratios, the system SO_2 removal was unaffected. It is likely that for the HCl concentration range tested, sufficient HCl was present to reduce the droplet drying time and enhance the dustcake reactivity by the formation and presence of calcium chloride to a point where higher HCl concentrations would not provide any further enhancement.

Shown in Figure 10 is the HCl system removal efficiency for all 79 test runs as a function of the spray dryer outlet temperature. Generally, the removal of HCl remained above 95 percent for most test conditions. In all cases, the stoichiometric ratio had very little effect on the HCl removal, except when the system was operated at stoichiometric ratios significantly below 1.0. The lower limit of the shaded area in Figure 10 represents operation at a stoichiometric ratio of about 0.5. Operating at a large excess of lime did not improve the HCl removal. The main factor controlling the HCl removal was found to be the spray dryer outlet temperature. At a spray dryer outlet temperature of 240 °F, the HCl removal averaged around 97 to 98 percent while at a spray dryer outlet temperature of 320 °F, the average HCl removal was about 94 to 95 percent. Generally, the HCl removal efficiency in the spray dryer and in the baghouse was comparable.

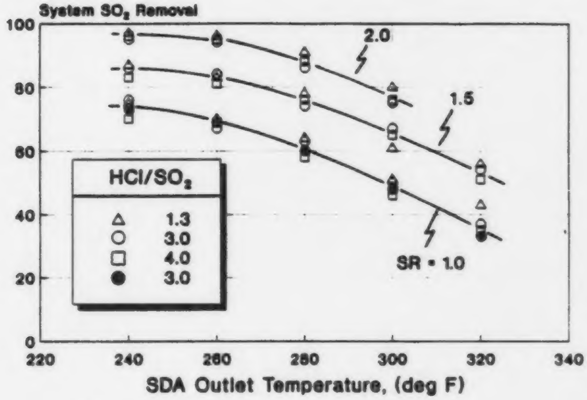

Figure 9. Influence of flue gas composition on system SO_2 removal.

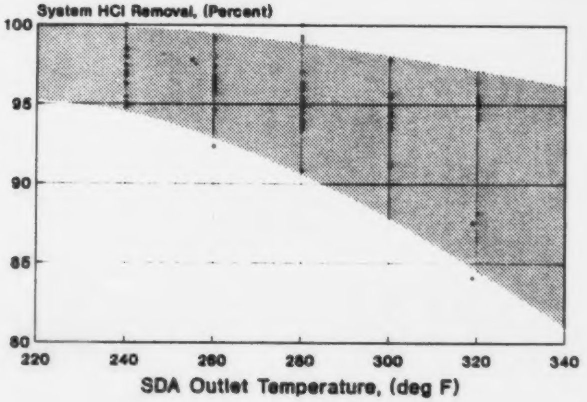

Figure 10. System HCl removal as a function of spray dryer outlet temperature. All test conditions included.

Recycle of the dry baghouse product is employed at most dry scrubbing systems applied on coal-fired boilers and the beneficial effect of recycle on the lime utilization has been well documented. To date, however, recycle has not been widely employed on MWC facilities. Figure 11 illustrates the effect of ash recycle on the system SO_2 removal efficiency; even modest recycle ratios were found to greatly enhance the removal of SO_2 while maintaining the HCl removal efficiency.

The results from the recycle test runs displayed in Figure 11 were obtained at a stoichiometric ratio of 0.5. Hence, without recycle, the SO_2 removal efficiency was limited to 50 percent, as shown in Figure 11. With recycle, the SO_2 removal efficiency increased dramatically and seemingly exceeded the theoretical limit. However, a test run using only slurried MSW fly ash as a sorbent revealed that when slurried, MSW fly ash can remove up to 50 percent of the HCl while the SO_2 is not affected. Hence, during the recycle tests, a fraction of the HCl was removed by the ash leaving lime otherwise used for removal of HCl available for SO_2 removal. In fact, assuming 50 percent HCl removal by the MSW fly ash, the apparent reduced stoichiometric ratio during the recycle test was about 1.3.

At no time did recycle present any problems with drying or material handling of the baghouse product. This is not surprising as on a mass basis, recycle does not increase the calcium chloride concentration of the dry baghouse product. Rather, recycle acts to reduce the drying rate of the droplets because of the presence of calcium chloride in the slurry from the time the slurry is atomized. Without recycle, the presence of calcium chloride must be obtained through absorption of HCl and part of the drying time is consumed before a saturated solution of calcium chloride is established.

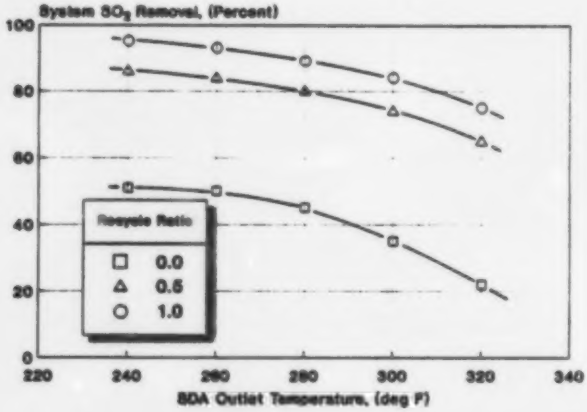

Figure 11. Effect of recycle on system SO_2 removal. Stoichiometric ratio 0.5.

BAG MATERIAL

Membrane laminated filter bags are inherently capable of very high particulate removal efficiencies and hence, are particularly suitable where very stringent particulate emission regulations apply. A membrane bag has a fine membrane laminated to a felt or a fabric backing. Effective filtration does not depend on the presence of a dust cake on the bag but takes place on the membrane surface. The membrane is commonly made of a smooth, non-sticky P.T.F.E. (polytetraflourethelene) material, which in combination with surface collection rather than cake filtration, reduces the dust cake on the bag to an absolute minimum.

To study the influence of the bag material on the removal of HCl and SO_2, the conventional Teflon® coated fiberglass bags used previously, were replaced with membrane laminated fiberglass bags (GORE-TEX® filter bags). The results from the tests are shown in Figure 12. It was determined that for all practical purposes, conventional woven fiberglass bags and membrane laminated fiberglass bags are comparable in terms of SO_2 and HCl removal. As the membrane laminated bags did not carry any dust cake in conventional terms, this further supports that it is only the surface or outer layer of the dust cake that is active in removing HCl and SO_2 while the bulk of the cake has no or only a marginal capability for HCl and SO_2 removal. The particulate removal capabilities of the membrane laminated bags were superior but will be reported elsewhere.

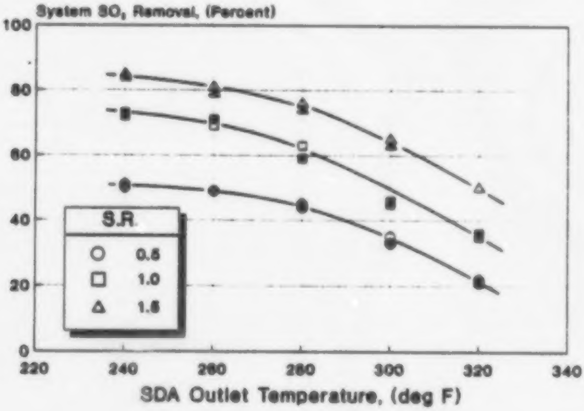

Figure 12. Comparison of the performance of conventional Teflon coated fiberglass bags (open symbols) and membrane laminated fiberglass bags (closed symbols).

ASH CHARACTERIZATION

The municipal fly as used in this study was obtained from the Tuscaloosa Waste Authority by permit from the US EPA/ADEM under the authority of the Resource Conservation and Recovery Act. The fly ash was generated from the mass burn of municipal waste in four 75-ton per day modular incinerators using one common two-field electrostatic precipitator for control of particulate emissions. The fly ash was classified as hazardous according to the EPA Extraction Procedure (EP) toxicity test due to high concentrations of lead and cadmium in the leachate. The EP-toxicity leachate concentration for the raw fly ash averaged 11 ppm for lead and 29 ppm for cadmium. Currently, the limit for lead is 5 ppm while for cadmium it is 1 ppm. The concentration of other heavy metals did not exceed the EP-toxicity limits. The raw fly ash was about 43 percent soluble by weight in distilled water. The major anions were determined to be sulfate and chloride, with the majority being sulfate. The pH of the raw fly ash was determined to be 5.6 by using EPA Method 90-45. An ion chromatograph was used to analyze heavy metals in the raw ash and in the baghouse end product.

The baghouse residue was tested for heavy metals leachate using the EP-toxicity method. Lead and cadmium concentrations in the leachate generally were reduced an order of magnitude from those in the raw fly ash. Although some trends were observed, no clear relationships among experimental conditions and metal leachate concentrations were determined. However, the ratio of lime to raw fly ash was found to affect the concentration of cadmium. Whereas the concentration of lead and cadmium in the leachate generally decreased a factor of ten with addition of lime, the effect does not appear to be simply dilution. With a fly ash loading of 2 grains/acf (lime to fly ash ratio of 1:6), the cadmium concentration in the leachate averaged 2.32 ppm while at a fly ash loading of 1 grain/acf (lime to fly ash ratio of 1:3) the cadmium concentration was 0.39 ppm. There was no significant difference in the lead leachate concentration with the fly ash mass loadings (1.19 ppm versus 1.16 ppm). Hence, at higher lime to fly ash ratios, the baghouse residue passed the EP-toxicity test and could be rendered as non-hazardous. The pH of the baghouse residue was about 8.0 as determined by EPA Method 90-45. Thus, the resulting effect of lime addition is more likely due to the reduced solubility of lead and cadmium compounds at the higher pH.

Recycling the baghouse residue resulted in leachate concentrations of lead about twice that of the raw fly ash and cadmium concentrations of less than 1 ppm (19 ppm lead and 0.16 ppm cadmium). Not enough data is available to determine the cause of this. The additional removal of SO_2 and the interaction between the ash and HCl by recycling of baghouse residue may have caused complexing of cadmium to less soluble forms (such as hydrated cadmium sulfates), while at the same time shifting the lead compounds to more soluble forms or ones that favor reaction to lead acetate when contacted with acetic acid. The complex chemistry involved in this phenomena needs to be further investigated.

DISCUSSION

The test results presented above support the assertion that spray dry absorber/fabric filter systems are among the best available technologies for control of acid gases and particulate emissions from MWC facilities. Dry scrubbing systems are capable of consistently achieving high HCl and SO_2 removal efficiencies and meeting stringent emission standards. HCl removal efficiencies above 95 percent are commonly obtained while, depending on operating conditions, SO_2 removal efficiencies above 90 percent are feasible.

Furthermore, the pilot plant study quantified the effect of important operating parameters on the system performance, including spray dryer outlet temperature, stoichiometric ratio, spray dryer inlet temperature, and the concentration level of HCl and SO_2. It was clearly demonstrated that operating at a close approach to the calcium chloride dew point is feasible and greatly beneficial to the lime consumption. As a rule of thumb, for a given SO_2 removal efficiency, the stoichiometric ratio would have to be doubled if the SDA outlet temperature was increased from 240 to 300 °F. It was demonstrated that spray dry scrubbing system can be operated down to 220 °F without any material handling problems. This agrees well with Combustion Engineering's experience with on-line commercial units that can and do operate down to 230 °F. At no time did operation at a close approach create any problems with handling of the baghouse product.

Operating at a close approach to the calcium chloride dew point requires attention to the selection and design of some key components of the dry scrubbing process. Slurry atomization and gas/liquid mixing is the most important unit operation, particularly when applied on MWC facilities. Rotary atomization provides the fine and uniform spray necessary to assure complete drying and to avoid buildup in the dryer. Furthermore, rotary atomizers provide consistent performance and any deterioration of the spray during operation is unlikely. Rather, if a problem with the spray machine occurs, built-in control functions will take the spray machine out off service to avoid uncontrolled buildup in the dryer. Furthermore, operating at a close approach requires properly insulated hoppers, hoppers have to be adequately heated and hopper area leakage kept to a minimum.

Equally important is good liquid and gas mixing, with respect to both drying and avoiding buildup and to obtaining optimum absorption conditions. For this purpose, the slurry-to-gas momentum ratio must be properly selected. A high slurry-to-gas momentum ratio would produce a spray cloud shape which rises and impinges on the ceiling and high wall surfaces, while a low ratio can produce a cloud shape which is quite narrow and non-conducive to gas/liquid contact. Keeping the gas-to-liquid momentum ratio at a reasonably constant level during turndown is equally important. To accomplish this, Combustion Engineering has developed the patented 40/60 spray dryer inlet design which can be applied in high turn-down applications (1).

The pilot study further revealed the benefit of using recycle on lime consumption. Recycle has the potential of reducing the lime consumption between 30 to 50 percent depending on the flue gas composition and operating conditions. Opposite to wet scrubbing systems operating in a closed loop configuration, recycle of baghouse solids does not on a mass basis increase the chloride level in a dry scrubbing system. Hence, systems employing recycle can operate at the same spray dryer outlet temperature as systems designed for once through operation. The material selection for systems employing recycle needs to reflect the corrosive nature of a slurry holding significant levels of chlorides. Furthermore, the dry recycle stream would require heating to avoid condensation and adsorption of water by the calcium chloride in the ash, particularly in the intermediate storage silo for recycle ash.

The choice of bag material (dust cake) was determined not to effect the removal of SO_2 or HCl. This is in agreement with the finding that the baghouse pressure drop (dust cake thickness) does not affect SO_2 or HCl removal.

The baghouse residue was tested for heavy metal leachate using the EP-toxicity method. Although no clear trends among experimental conditions and metal leachate concentrations could be established, it was determined that the concentration of lead and cadmium generally was reduced an order of magnitude from those in the raw ash. It was further determined that at high lime feed rates (lime to fly ash ratio of 1:3), the baghouse residue becomes non-hazardous.

ACKNOWLEDGEMENT

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COMPARISON OF PERFORMANCE OF SYSTEM CONFIGURATIONS OF POLLUTION CONTROL SYSTEMS FOR MSW INCINERATION

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ABSTRACT

Emissions resulting from incineration have been treated via alternate systems designed for abatement of acid gases, particulates, heavy metals, and organics including dioxins and NO_x. System selection is based on regulatory requirements, volume of gas treated, skill of personnel, and whether the system is green field or retrofitted to existing particulate collection equipment.

The Teller Process has been commercially operating in the following configurations :

Quench Reactor - Dry Venturi - Fabric Filter
Quench Reactor - Fabric Filter
Quench Reactor - Dry Venturi - Precipitator
Quench Reactor - Precipitator
Dry Venturi - Fabric Filter

Noxious-toxic component removal efficiency, type of reagents and consumption, operational characteristics, reliability and economics are compared for 12 typical operating systems.

INTRODUCTION

The Research-Cottrell/Teller (RC/Teller) system is in operation, providing emission controls, on over 100 incinerator trains. Applications include Resource Recovery, Hazardous Waste, and Hospital Waste installations.

Five configurations of the system have been utilized. These include several different components integrated into an overall system. System configurations include:

Quench Reactor - Dry Venturi - Fabric Filter (QR-DV-FF)
Quench Reactor - Fabric Filter (QR-FF)
Quench Reactor - Dry Venturi - Precipitator (QR-DV-ESP)
Quench Reactor - Precipitator (QR-ESP)
Dry Venturi - Fabric Filter (DV-FF)

The first four configurations listed are considered to be "semi-dry". That is, they contain a reaction vessel into which is injected a lime slurry for initiating the neutralization of acid gases. The configuration of a Dry Venturi with a Fabric Filter is an "all-dry" system. Dry, hydrated lime is used as the reagent for acid neutralization. Figure 1 shows a process schematic for the configuration which includes all of the different components.

The selection of the system configuration is based upon the regulatory requirements, process considerations and economic factors.

The range of conditions serviced by these systems is summarized in Table 1.

TABLE 1. RANGE OF SYSTEM INLET CONDITIONS

Incinerator capacity, TPD	10 - 1000
Flue Gas Conditions	
Humidity, (V/V) %	7 - 42
Temperature, ° C	175 - 925
Particulates, g/Nm^3	1 - 30
HCl, ppmdv	100 - 1500
SO_2, ppmdv	30 - 500
HF, ppmdv	1 - 50
Mercury, ug/Nm^3	50 - 10000
Dioxins, ng/Nm^3	5 - 30

The effectiveness of the reduction in emissions is a function of system configuration, humidity of the flue gas, operating temperature, the ratio of the reagent feed rate to the stoichiometric requirement, and the final collector type and cleaning cycle.

SYSTEM COMPONENTS

Each of the components included as part of the emission control system is considered a reaction vessel. As such, each plays an integral part of the overall system efficiency and reliability. The overall system is a combination of reactors, in series. Included are:

Initial Reactor -	Quench Reactor or Dry Venturi
Secondary Reactor -	Dry Venturi
Final Reactor -	Fabric Filter or Electrostatic Precipitator

QUENCH REACTOR

The Quench Reactor provides the first stage of neutralization of acid gases in the semi-dry process. This is accomplished via contact of the acid gases with lime slurry introduced through spray nozzles. The HCl neutralization occurs through rapid volatization of the water, followed by nucleation, as well as the interaction of HCl and water vapor. A fine mist of HCl is formed. This mist of HCl is contacted by the wet lime particles, and is neutralized in the liquid phase.

SO_2 is neutralized by diffusion of the SO_2 to the surface of the wet lime particle. The SO_2 absorption downstream of the Quench Reactor is further enhanced by the formation of $CaCl_2$ from the reaction of HCl and lime. This is due to the hygroscopic nature of the $CaCl_2$. Water is retained in the lime reaction product matrix, thus enhancing the kinetics for further absorption of SO_2.

The Quench Reactor is of proprietary upflow design (1). This provides a greater residence time for drying of the droplets than the gas residence time. The volume of the vessel is approximately 50 % smaller than that required for a downflow orientation for the same drying time.

The Quench Reactor incorporates a cyclonic inlet which removes a portion of the incoming particulates. Particulates which are larger than 150 microns are removed from the gas stream before it enters the reactor. Damage to downstream equipment due to sparks or embers which may enter the emissions control system is reduced.

DRY VENTURI

The Dry Venturi is a proprietary contacting device located in the duct upstream of the final particulate removal equipment (2). Tesisorb, as well as reagents can be injected into the Dry Venturi via a dilute phase pneumatic conveying system.

The Dry Venturi provides two services for the process. First, it serves as a fabric filter cake modifier, through the injection of Tesisorb. Submicron particulates are captured by Tesisorb due to the inertial impact and turbulent mixing in the Dry Venturi. As a result, the population density of the fine particulates is reduced. This results in a modification of the particle size distribution entering the fabric filter. The blinding potential of the filter bags is reduced and the cleaning cycle of the fabric filter is extended. The cleaning cycle can be extended up to 8-24 hours, with a lower pressure drop across the fabric filter. The extended cleaning cycle also increases the life of the filter bags. In one operating facility, the original filter bags installed eight years ago are still in service.

The second function of the Dry Venturi is for additional neutralization of acid gases. This occurs as a result of the turbulent mixing of the acid gases with reagent, present in the flue gas as it passes through the venturi.

The Dry Venturi serves as the initial reactor in the all-dry system (DV-FF). Neutralization of acid gases is initiated with the injection of dry hydrated lime.

FINAL COLLECTOR

The final component in the acid gas cleaning system is the particulate collector. This can be either a Fabric Filter (Baghhouse) or Electrostatic Precipitator.

The Fabric Filter is the preferred final control device in the acid gas cleaning train. Virtually all of the particulates entering the Fabric Filter are removed in the filter bags. The Fabric Filter also serves as a fixed bed reactor for the neutralization of acid gases. The long cleaning cycles allow residual reagent on the bags to react with acid gases entering the filter.

Both Pulse Jet type and Reverse Air type Fabric Filters have been incorporated as the particulate removal device.

The Electrostatic Precipitator has also been utilized as the final particulate collector in incinerator applications. The precipitator is used primarily as a particulate removal device. The precipitator is less effective than the fabric filter for acid gas neutralization.

Many of the applications which include the precipitator are retrofits.

FUNCTIONAL PARAMETERS

The effect of several key parameters on the removal efficiency of noxious-toxic compounds achieved in the various system components are described below. Operating temperature and moisture content, reagent characteristics in the Dry Venturi, fabric filter cleaning cycle, and electrostatic precipitator particulate collection are discussed.

The impact of the more obvious parameters, such as reagent stoichiometry, have been well documented and are discussed only in conjunction with other functional parameters.

EFFECT OF OPERATING TEMPERATURE AND MOISTURE CONTENT

The moisture content and the temperature of the gas stream are critical in establishing the efficiency of acid gas, dioxin and mercury removal.

In a "semi-dry" system, the efficiency of acid gas collection increases with a decrease in the operating temperature. Figure 2 illustrates the effect of flue gas temperature on the removal efficiency of HCl. These efficiencies are plotted for a range of reagent stoichiometric rates, or Alpha[1]. These curves are for a system which consists of a QR-DV-FF.

As the temperature of the flue gas is reduced, the efficiency of acid gas removal is enhanced at each level of Alpha. The effect of the temperature reduction diminishes, however, at levels of Alpha above 2.5. These Alpha levels are generally uneconomical for large capacity systems.

The effect of temperature on dioxin removal has also been demonstrated (3). The range of 120-140 °C has been found to be the optimum range to maximize the capture of dioxins in an acid gas cleaning system which includes a fabric filter.

Mercury removal is also affected by temperature. However, this effect is also dependent on the quality of the adsorbent in the gas stream. At the Skovde incinerator, an adsorbant was injected into the Dry Venturi to reduce mercury emissions. Emission levels of mercury were reduced from a level of 10,000 ug/Nm^3 down to 25 ug/Nm^3 at 140 °C (4).

Figure 3 shows the effect of moisture content on the removal efficiency of HCl. In this system, which is all-dry, the effect of moisture content is quite pronounced. At a constant Alpha=2.0, the HCl removal efficiency increases with moisture content, up to a moisture content of 20 %. Beyond this level the effect of moisture is diminished.

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[1] Alpha denotes the ratio of neutralization agent used to the stoichiometric requirement based on entering acid gas quantities.

EFFECT OF REAGENT CHARACTERISTICS IN THE DRY VENTURI

The Dry Venturi in the RC/Teller System also serves as reaction vessel for the neutralization of acid gases. The degree to which reactions occur in the Dry Venturi is a function of the characteristics of the reagent in the flue gas stream.

The reaction rate is enhanced if prior neutralization of HCl, by lime, has occurred, for example in a Quench Reactor upstream of the Dry Venturi. The reaction product, $CaCl_2$, generated as a result of HCl reaction, is deliquescent and adsorbs water from the surrounding flue gas. The adsorbed water, as well as water generated in the neutralization process, is held onto the lime particles. This results in an increase in the removal efficiency. Figure 4 shows the effect on HCl removal efficiency of the presence of $CaCl_2$. Improvement in the removal efficiency of HCl by the presence of $CaCl_2$ is shown, up to a mole ratio of $CaCl_2$ formed to $Ca(OH)_2$ fed (to the system) of about 2.

EFFECT OF FABRIC FILTER CLEANING CYCLE

The fabric filter is used as the final particulate removal device in the air cleaning system. It also acts as a final reaction vessel for acid gas removal. In the all dry system, the fabric filter is considered the principal reactor for acid gas neutralization.

The removal efficiency of acid gases in the fabric filter depends upon the operating temperature, flue gas moisture content, air to cloth ratio, cleaning cycle and the quantity of unreacted reagent in the cake on the filter.

Table 2 shows the effects of the cleaning cycle at several levels of Alpha on the HCl removal efficiency for a Quench Reactor-Baghouse system. This system operates at an air to cloth ratio of 64 $Nm^3/hr/m^2$ with a 15 % flue gas moisture content and temperature of 135 ^oC.

TABLE 2. EFFECT OF CLEANING CYCLE ON HCl REMOVAL EFFICIENCY FOR A QR-FF SYSTEM

HCl Removal Efficiency

	Cleaning Cycle :			
Alpha	10 min.	20 min.	1 hour	2 hour
1.2	57 %	66 %	81 %	91 %
1.4	63	71	87	95
1.6	69	76	91	97
1.8	73	81	94	98
2.0	78	85	96	98
2.2	81	87	97	98

While the values tabulated are overall efficiencies across the Quench Reactor and Fabric Filter, it is evident that the cleaning cycle has a significant effect. At any specific level of Alpha, the increase in HCl removal efficiency is attributed to cleaning cycle. A similar increase in HCl removal, due to an increase in the amount of lime entering the baghouse, is observed for each cleaning cycle.

Test data from these systems shows that HCl removal, in the fabric filter alone, can exceed more than 95 %. Figure 5 shows the baghouse efficiency as a function of residual Alpha, air to cloth ratio, and cleaning cycle, at a constant temperature and humidity.

A similar effect on the removal efficiency for SO_2 has been observed for both of the paramaters shown in the table.

ELECTROSTATIC PRECIPITATOR

The removal efficiency of acid gas measured across an electrostatic precipitator is lower than that for a fabric filter. The nature of the collector, as well as the frequent cleaning, result in reduced collection efficiencies of acid gases. The removal efficiency of acid gases for this type of collector is more dependent on the amount of reagent entering the precipitator and less dependant on cleaning cycle.

Figure 6 shows the effect of Alpha on the removal efficiency of HCl and SO_2 in the precipitator. Inasmuch as the absorption-reaction time is a constant for the precipitator, the efficiency is a function of the residual Alpha.

SYSTEM PERFORMANCE

Table 3 summarizes operating data for the various configurations of the Research-Cottrell/Teller System operating at several incinerator installations. The performance of the system configurations at these sites is tabulated in Table 4. Inlet and outlet emissions data and removal efficiencies are presented.

TABLE 3. OPERATING DATA FOR INCINERATOR INSTALLATIONS

Configuration	Installation	TPD	Start Year	Flow NCMH	Temp °C	H_2O %
QR-DV-FF	Tsushima	165	1983	29890	360	24
	Marion	275	1986	55890	205	16
	Commerce	330	1986	93380	261	15
QR-FF	Jackson	100	1987	24830	178	16
	Skagit	90	1988	23050	246	13
DV-FF	Tsushima	165	1983	29890	360	24
	Skovde	25	1987	9200	145	7.5
	Dutchess	200	1988	50360	210	N/A
QR-ESP	Shizuoka	250	1982	46500	290	24
	Sakai	300	1980	51550	326	23
	Kamakura	165	1982	30420	370	24
QR-DV-ESP	Fujisawa	165	1987	51290	321	45

TABLE 4. PERFORMANCE DATA FOR INCINERATOR INSTALLATIONS

		SYSTEM INLET			SYSTEM OUTLET			EFFICIENCY	
Configuration	Installation	PART mg/Nm^3	HCl ppm	SO_2 ppm	PART mg/Nm^3	HCl ppm	SO_2 ppm	HCl %	SO_2 %
QR-DV-FF	Tsushima	4580	313	13	27	7.5	0.04	97.6	99.6
	Marion	2900	470	115	21	9.5	7.6	97.8	93.4
	Commerce	4280	730	250	10	11	2	98.5	99.2
QR-FF	Jackson	2280	730	91	7	22	15	97.0	83.5
	Skagit	1370	311	169	6	10	N/A	96.8	N/A
DV-FF	Tsushima	4580	250	40	N/A	15	16	94.0	60
	Skovde	280	1220	N/A	2	12	44	99.0	N/A
	Dutchess	N/A	N/A	60	9	N/A	20	N/A	67
QR-ESP	Shizuoka	N/A	1000	90	N/A	150	40	85.0	55.5
	Sakai	N/A	334	38	5	34	9	89.8	76.5
	Kamakura	N/A	270	80	N/A	30	21	88.8	73.7
QR-DV-ESP	Fujisawa	N/A	667	61	N/A	49	9	92.6	85.2

Figure 7 shows the SO_2 emissions, based on 24 hour averages, measured at the City of Commerce. These data were collected from January through December 1988.

Recent data for SO_2 emissions at this facility are presented in Table 5. These data were collected after modifications to the reagent feed system were completed. These modifications were made to permit reagent feed controlled on a signal from SO_2 monitors. Previously, the control of reagent feed was manually adjusted. The data summarized in Table 5 were collected during the period of February 1-5, 1989.

TABLE 5. RECENT EMISSIONS DATA FROM THE CITY OF COMMERCE

Date	SO_x lb/day	SO_x lb/hr.
2/1/89	4.4	0.18
2/2/89	21.3	0.89
2/3/89	34.5	1.44
2/4/89	14.8	0.61
2/5/89	18.4	0.77

Performance of the various configurations with respect to removal of heavy metals and organic compounds have been presented previously and are not discussed in this paper (4)(5).

ECONOMICS

The economics for the various configurations of acid gas cleaning systems have been developed for a hypothetical 200 TPD MSW incinerator. The system is based on a total turnkey installation, from the economizer outlet to the stack inlet, for a single boiler. All of the equipment required to operate the system, with the exception of ash handling equipment, are included as part of the installed cost. The system ID fan is also included in the capital and operating cost analysis. A pulse jet fabric filter is used in this example.

The design of the emissions control system is based on typical incinerator inlet conditions. These include:

Boilers	1 @ 200 TPD
Gas volume, ACFM	40,000
Gas temperature, ° F	400
Flue gas conditions:	
HCl, ppmdv	500
SO_2, ppmdv	150
HF, ppmdv	20

Table 6 presents the annualized costs for the various configurations as applied to the system described. These costs are presented on a \$/ton basis. The assumptions in this analysis are:

Amortization - 10 year straight line
Chemicals - CaO \$80/ton; $Ca(OH)_2$ \$100/ton; Tesisorb \$40/ton
Water - \$1.00/1000 gal.
Power - \$0.06/kWh
Operation - 8,200 hours/year (340 days)

The labor and maintenance costs are assumed values, based on operating experience at some of these plants.

TABLE 6. ANNUALIZED CAPITAL AND OPERATING COSTS

Item	QR-DV-FF	QR-FF	QR-DV-ESP	QR-ESP	DV-FF
Amortization	3.16	2.85	3.38	3.06	1.95
Chemicals and water	0.69	0.64	1.19	1.09	1.72
Power	1.37	1.17	1.59	1.39	0.70
Labor and maintenance	0.55	0.50	0.45	0.40	0.30
Total, \$/ton	5.77	5.16	6.61	5.94	4.67

As previously stated, the costs above are for a single incinerator gas cleaning train. For installations which include multiple incinerators, the amortization costs, on a per ton basis will decrease due to sharing of common equipment. This reduction will be more pronounced for the DV-FF system. This is because one of the major equipment costs associated with this system are for lime and Tesisorb storage silos, which would be shared among multiple trains.

The table illustrates some of the considerations involved in system selection. The all-dry system has the lowest capital costs but has a higher chemical cost. The choice of the final collector, precipitator or fabric filter, impacts both the amortization and chemical costs.

CONCLUSIONS

The direct comparison of the performance of the various configurations is dependent on many factors, including many of the functional parameters previously discussed. However, a pattern of performance is evident. Table 7 summarizes the performance for the various configurations. The annualized costs for the 200 TPD incinerator are also tabulated.

TABLE 7. PERFORMANCE CHARACTERISTICS OF SYSTEM CONFIGURATIONS

Configuration	Removal Efficiency, % HCl	SO_2	Heavy Metals	Annualized Costs $/ton
QR-DV-FF	95	95	99.7	5.77
QR-FF	90	85	99.7	5.16
DV-FF	90	60	99.7	4.67
QR-DV-ESP	90	70	99.0	6.61
QR-ESP	85	70	99.0	5.94

The performance for the various configurations can be directly related to the $/ton cost. For a system which includes a Fabric Filter as the final collector, a higher level of performance is achieved at an increase in annualized cost.

The system configuration of QR-ESP can be designed to achieve greater than 70 % removal of SO_2 at high reagent stoichiometries.

The retrofit of a Quench Reactor and Dry Venturi to an existing incinerator which has a precipitator can be applied for acid gas emissions control to achieve 90 % HCl and 70 % SO_2 removal. Operating conditions for this design would be different than a QR-FF system.

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2. Teller, A.J. et al, U.S.Patent 4319890, 1982
3. Teller, A.J. et al, U.S.Patent 4502396, 1985
4. Teller, A.J. et al, "Emission Controls on a Hospital Waste Incinerator", to be published.
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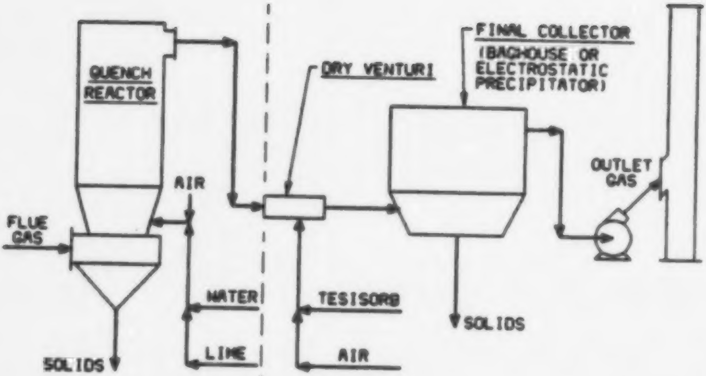
QUENCH REACTOR
FLUE GAS
AIR
WATER
LIME
SOLIDS
DRY VENTURI
TESISORB
AIR
FINAL COLLECTOR
(BAGHOUSE OR ELECTROSTATIC PRECIPITATOR)
SOLIDS
OUTLET GAS

FIGURE 1 - TYPICAL R-C/TELLER DRY SYSTEM

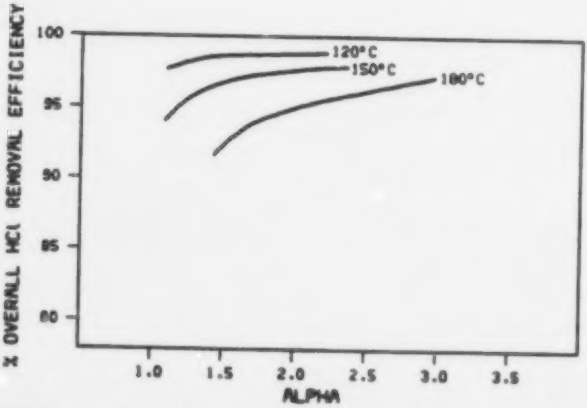

FIGURE 2 - EFFECT OF OPERATIONAL TEMPERATURE FOR HCl REMOVAL FOR A QR+DV+FF SYSTEM

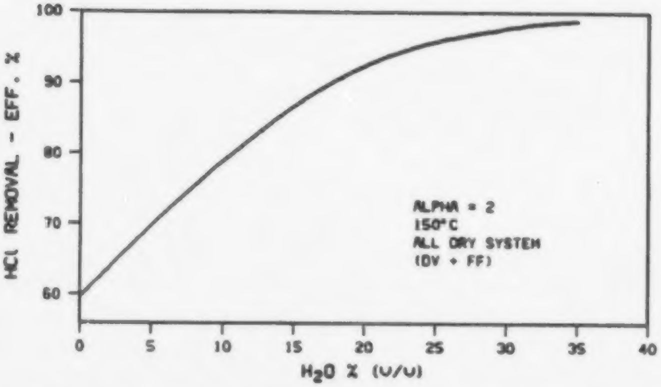

FIGURE 3 - MOISTURE EFFECT ON HCl REMOVAL

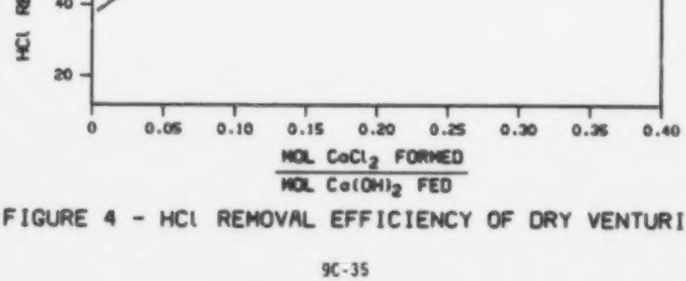

FIGURE 4 - HCl REMOVAL EFFICIENCY OF DRY VENTURI

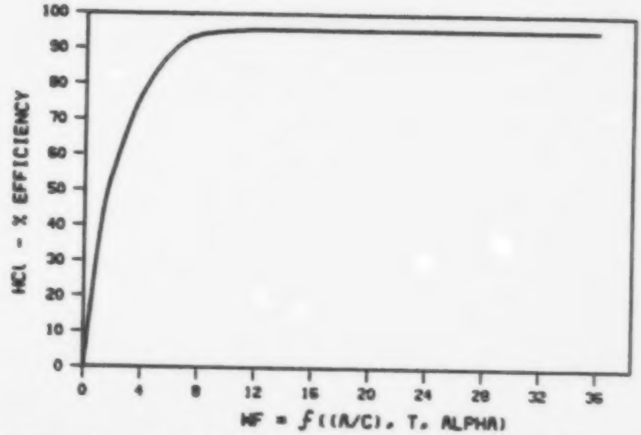

FIGURE 5 - HCl REMOVAL EFFICIENCY OF FABRIC FILTER

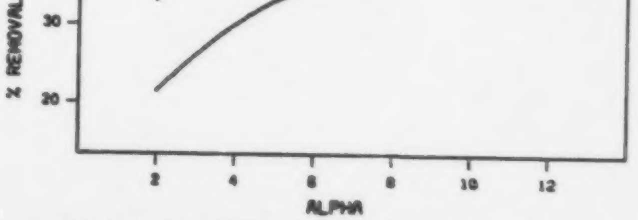

FIGURE 6 - REMOVAL EFFICIENCY FOR ESP

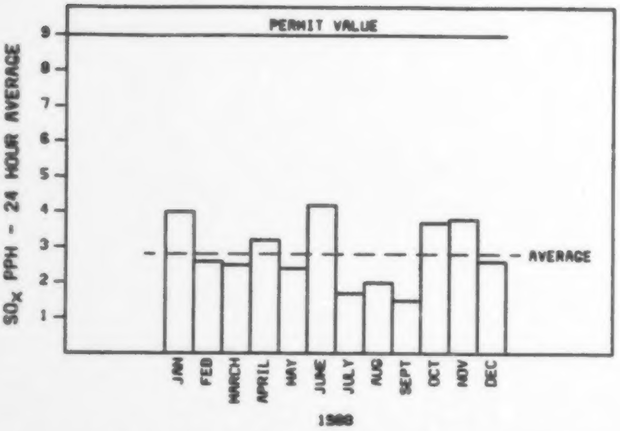

FIGURE 7 - SO_2 EMISSION - CITY OF COMMERCE

JOY/NIRO SDA-FGC SYSTEMS

NORTH AMERICAN AND EUROPEAN OPERATING EXPERIENCE

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ABSTRACT

Incineration of municipal solid waste (MSW) as a method of reducing disposal volume requirements and recovery of energy has been practiced extensively for many years in Europe, and more recently in North America. Concerns over potential air pollution have resulted in the promulgation of emissions standards for a wide range of pollutants and the application of spray dryer absorption (SDA) flue gas cleaning systems to control incinerator emissions. In Europe, many SDA applications have been retrofitted on existing incinerators equipped with electrostatic precipitators (ESP) for dust collection. Emissions standards have required stringent control of acid gases, particulate matter, and trace metals. SDA systems have demonstrated the ability to achieve required levels of control with either an ESP or fabric filter dust collector.

In comparison, only a relatively few incinerators equipped with SDA flue gas cleaning systems have started-up in North America. Data collected during initial testing of these systems are being used as a basis for setting emission requirements for plants currently in the permitting process. Little data is currently available on the performance of SDA systems which incorporate electrostatic precipitators as dust collectors.

This paper presents a comparison of emission requirements for JOY/Niro SDA systems in operation or construction in Europe and North America. Design and performance test data for European SDA systems equipped with either ESP's or fabric filters are given. Operating data are presented for new North American units equipped with ESP's.

INTRODUCTION

Incineration of municipal solid waste (MSW) has been utilized for many years in Europe and more recently in North America to substantially reduce the volume of waste to be disposed of while at the same time producing energy in the form of steam or electricity. As the application of incinerators has increased concerns have been raised over potential air pollution impacts associated from their use. This has lead to the promulgation of emissions limits and control requirements for a wide range of pollutants, first in the European countries, and more recently in the states and provinces in North America. Spray Dryer Absorption (SDA) flue gas cleaning systems have been successfully applied to control incinerator air pollutants.

Joy Technologies Inc. and A/S Niro Atomizer have been jointly developing SDA gas cleaning systems since 1977 and have successfully applied this technology for control of acid gases, particulates and trace element emissions from coal fired boilers, mass burn and RDF fired MSW incinerators and hazardous waste incinerators. Today more than 23 MSW incinerator trains are in operation employing this technology with an additional 19 trains under construction. The majority of the operating units have successfully completed performance tests and operate reliably while maintaining the incinerators in compliance with local air pollution regulations. This success can be attributed to Niro's vast knowledge of spray drying and Joy's extensive experience in application of particulate control technologies to a wide range of processes.

The systems applied in Europe and North America while utilizing the same basic technolgy incorporate many different design features for each particular application. The majority of the systems installed in Europe were retrofitted to existing incinerators where increasingly more stringent air pollution regulations require acid gas and trace metal emission controls on both new and existing plants. These systems (where possible) utilize existing electrostatic precipitators either as the main dust collector downstream of the spray dryer or as a precollector upstream of the spray dryer. A few of the newer installations employ a fabric filter as the dust collector.

In North America the opposite is true. The majority of SDA applications are on new incinerators. Whereas the first four Joy/Niro systems to start-up in the United States have electrostatic precipitators as dust collectors, the remaining 13 trains under construction are equipped with fabric filters.

This paper presents a description of the Joy/Niro SDA process and addresses key design parameters for successful applications. Design parameters and emission requirements for European installations are presented. Performance test data is given for the Kassel installation, the first Niro MSW flue gas cleaning system to employ a fabric filter as the dust collector. Performance test data from two installations at Leverkusen, Germany, is summerized. North American design parameters are presented and initial operation of the SEMASS system is discussed.

SDA SYSTEM DESCRIPTION

The spray dryer absorption (SDA) flue gas cleaning system as developed by Joy and Niro employs a spray dryer to evaporate a feed reagent to dryness while simultaneously removing acid gases from a flue gas stream by reaction with alkali in the feed. A dust collector follows the spray dryer to collect the dried reaction products and to act as a secondary reactor for further acid gas and trace metal removal. Key components of the process are the spray dryer, the dust collector and the reagent preparation system. Typically, one spray dryer and one dust collector are furnished per incineration train, although Joy/Niro spray dryers can be sized to handle the total flue gases from multiple incinerator trains. The reagent preparation system is typically sized to supply reagent to however many spray dryers are utilized. Figure 1 is a simplified flow sheet for the process.

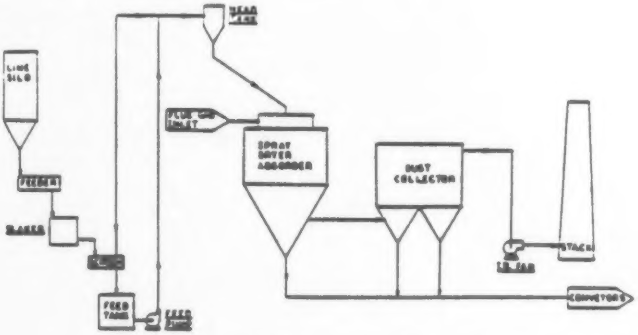

Figure 1. Joy/Niro Process Flow Sheet.

Pebble lime (CaO) is normally used as the reagent for acid gas applications although pulverized lime, hydrated lime [$Ca(OH)_2$] or sodium carbonate can also be used. Lime is delivered by truck and conveyed pneumatically to a storage silo. From here lime is fed to a slaker where a 20-25% hydrated lime slurry is prepared. The lime slurry is either stored in a lime milk tank or transferred directly to a feed tank. The lime slaking system is typically furnished as a complete package including silo, feeder, slaker, storage tank, transfer pumps and instrumentation. Either detention or paste slakers are used. The lime slurry is screened and then stored in the feed tank. From the feed tank, the lime slurry is pumped via a recirculation loop to a head tank. Lime is added to the head tank based on a feedback signal from an acid gas concentration in the stack or a percent removal signal. Dilution water is added to the head tank to maintain a liquid level.

The diluted lime slurry flows to the atomizer through an SDA outlet temperature control valve. The spray dryer outlet temperature is directly proportional to the amount of water entering the atomizer assuming inlet gas conditions remain constant. A feed forward signal based on inlet flue gas flow and temperature corrects for changes in inlet conditions enabling the control system to maintain the SDA outlet temperature within about ±2°F. The lime slurry enters the spray dryer via the atomizer where it is converted into a fine droplet cloud with a high surface area.

As the lime slurry enters the spray dryer, the droplet cloud is mixed with the hot incoming flue gas, where lime simultaneously reacts with acid gases present and dries prior to exiting the spray dryer absorber. The majority of reaction products formed are carried over to the dust collector where additional acid gas removal takes place. The dry product collected in the SDA and dust collector is then conveyed mechanically or pneumatically to a waste product silo. The product can also be conveyed to the bottom ash quench tank for mixing with the bottom ash. If economically justified, a portion of the dry product can be recycled back to the feed system to improve reagent consumption and acid gas removal.

The dry product formed consists primarily of calcium chloride ($CaCl_2$), calcium sulfite/sulfate ($CaSO3/CaSO_4$), calcium carbonate ($CaCO_3$), unreacted lime [$Ca(OH)_2$] and fly ash. The product is hygroscopic due to its calcium chloride content. This property influences the design spray dryer outlet temperature.

The spray dryer outlet temperature is controlled at a temperature low enough for efficient acid gas absorption and to maximize condensation and collection of volitile trace elements present in the flue gas. At the same time the temperature is maintained high enough to ensure a dry, free-flowing product under all operating conditions.

The dust collector downstream of the spray dryer may be an electrostatic precipitator, a reverse-air baghouse or a pulse-jet type baghouse. The selection of a specific type of dust collector is dependent on site specific factors such as particulate emission limits, overall acid gas removal requirements, project economics and client preference. Each of these dust collection devices offers process advantages and disadvantages that should be evaluated on a site specific basis. Special features are incorporated into the design of the dust collector for operation downstream of the spray dryer.

SPRAY DRYING DESIGN PARAMETERS

Spray dryer absorption is a special application of spray drying technology where efficient drying must also be balanced with optimum reagent utilization. Both efficient spray drying and spray dryer absorption are dependent on a few key inter-related system design parameters. These parameters include:

- Atomization
- Gas Dispersion/Mixing
- Dryer Residence Time
- Dryer Outlet Temperature

ATOMIZATION

Atomization is the conversion of a liquid stream into a multitude of discrete fine droplets. This may be accomplished through the use of rotary atomizers or nozzles (pressure or dual fluid) depending on properties desired in the end product. For SDA applications, Joy/Niro have selected single rotary atomization for the following reasons:

- Proven high reliability in spray drying.
- Ease of changing atomization by changing atomizer speed.
- More energy efficient atomization.
- Simplicity of operating one atomizer compared to multiple nozzles.
- Proven long life of the patented KN abrasion resistant wheel.

The atomizers used for SDA applications have demonstrated greater than 95% availabilities in many industries handling abrasive feed slurries at rates both much higher and lower than for incinerator applications. The degree of atomization and the spray pattern are, to a great extent, insensitive to feed flow rates and wear. The degree of atomization is set by the peripheral speed of the wheel and can be optimized for each application by changing gear ratios. Atomization that is too fine can result in rapid drying and increased reagent consumption, whereas atomization that is too coarse can result in poor drying and chamber deposits. Typical wheel speeds range from 12,000 to 16,000 rpm for incinerator applications, depending on type of standard atomizer speed and wheel diameter combination selected. Droplet mean particle sizes range from 30 to 50 microns.

The use of a single, centrally mounted rotary atomizer provides for a uniform outlet temperature distribution and simplifies control of the spray dryer outlet temperature. The spray dryer outlet temperature is directly controlled by changing the feed rate to the atomizer and remains constant within a few degrees. This type of atomization allows for reliable low outlet temperature operation.

A rotary atomizer is more efficient and requires less power than nozzle atomizers to achieve the same degree of atomization for abrasive feed materials. When compared to the compressed air requirements for dual fluid nozzle atomization, energy savings of 30% can be realized in some applications. Figure 2, following page, shows a Niro rotary atomizer and KN abrasion resistant wheel. This patented wheel design was initially developed for handling very abrasive wet cement and mineral concentrate slurries at up to 55% solids.

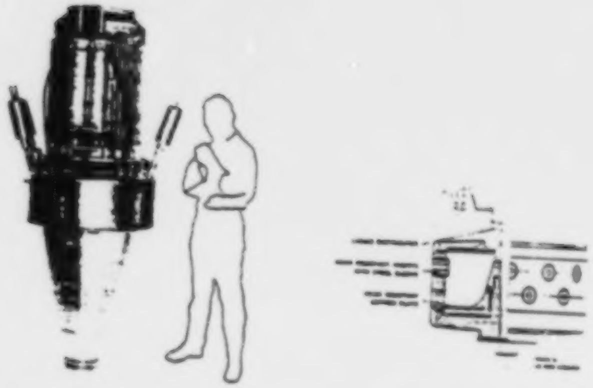

Figure 2. Rotary Atomizer and KN Wheel.

The wheel consists of a wheel body, made of stainless steel or more corrosion resistant alloy, and hardened inserts protruding into the wheel body to form a centrifuged layer of slurry solids for protection of the inside of the wheel body. The bottom plate and the inserts are made of abrasion-resistant material; e.g., silicon carbide. Wheel materials are selected based on expected acid gas concentrations, feed slurry solids content and SDA inlet flue gas temperature. They have demonstrated reliable long-term life in these applications.

GAS DISPERSION

Intimate contact between the drying media (flue gas) and the atomized droplets (lime feed slurry) is required for rapid heat and mass transfer. The amount of energy required to achieve this mixing varies dependent on chamber size, turndown requirements, chamber geometry and allowable spray down temperature (temperature drop across the spray dryer). The gas disperser controls the shape of the atomized droplet cloud, provides efficient mixing of hot gas and atomized slurry in a narrow zone around the atomizer, and rapidly reduces the flue gas temperature to close to the outlet temperature.

A roof gas disperser is used for municipal solid waste incinerator applications where it is desirable to operate at low spray dryer outlet temperature. Figure 3 shows a spray dryer equipped with a roof gas disperser.

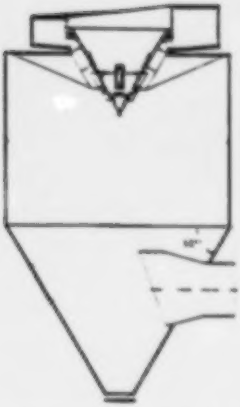

Figure 3. Spray Dryer with Roof Gas Disperser.

This type of disperser can maintain efficient gas/droplet mixing from approximately 25 to 110% of design flow at low pressure drops. The roof gas disperser is equipped with sets of adjustable vanes for optimization of mixing. These vanes are shown in Figure 4.

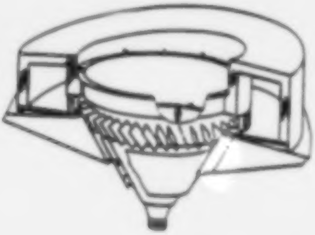

Figure 4. Roof Gas Disperser Vane.

During initial start-up the vanes are set to impart a swirl to the incoming flue gas thus controlling the shape of the atomized droplet cloud.

DRYER RESIDENCE TIME

Spray dryer flue gas residence time is selected to ensure adequate drying time before the atomized slurry droplets impact on the chamber walls or the outlet duct. Minimum retention time is a function of degree of mixing, atomization, feed solids composition and concentration, outlet gas temperature and expected flow variations. The minimum residence time must be sufficient to dry the largest atomized droplets before they impact on the spray dryer walls to avoid wall deposits. For example, in utility applications using a compound gas disperser and product recycle where the atomizer feed contains approximately 40 to 50% solids, a nominal 10 seconds residence time is adequate to dry at 18-20°F above the adiabatic saturation temperature.

For incinerator applications the fluctuation in flue gas flow rates is more rapid and greater than utility applications. In addition, the feed solids concentration may vary from 3 to 20%. These factors result in increased residence time required to ensure adequate drying under all operating conditions. Typically, the nominal residence time is maintained between about 15 and 21 seconds when the outlet temperature is maintained between 240-275°F (100-130°F above the adiabatic saturation temperature).

DRYER OUTLET TEMPERATURE

The spray dryer absorber outlet flue gas temperature should be controlled as low as possible to maintain a high flue gas relative humidity thus resulting in maximum reagent utilization and absorption efficiency. The temperature must, at the same time, be maintained high enough to ensure that the product is dry and free-flowing. Primary factors which affect the minimum safe outlet temperature are the nature of the dried product, the feed solids concentration and the degree of gas/slurry mixing. For incinerator applications, the product is somewhat hygroscopic due to the presence of calcium chloride. The hygroscopicity varies proportionally with the chloride content.

For single pass operation, the outlet temperature is normally controlled between 240-275°F; whereas with product recycle, the temperature is maintained somewhat higher due to increased calcium chloride concentration in the product. In applications where a precollector is used in front of the spray dryer the product chloride concentration may reach 50%. For these applications an outlet temperature of 270-285°F is selected.

EUROPEAN APPLICATIONS

Niro has supplied 14 SDA systems for European MSW incinerators since 1986. These systems are almost exclusively retrofit applications that utilize ESP's as the dust collector. All of the systems except those in France have removal requirements for trace metals, as well as for HCl, SO_2, and HF. Design conditions for these systems are summarized in Table I.

TABLE 1. EUROPEAN DESIGN CONDITIONS

TYPE WASTE	MUNICIPAL SOLID WASTE							
PROJECT	City of Zurich, Switzerland	City of Leverkusen, W. Germany	AVG Hamburg, W. Germany	City of Nice, France	City of Solingen, W. Germany	City of Kassel, W. Germany	Amager-forbraending Denmark	Carriere Sur Seine France
Inlet Flue Gas to SDA								
Flue Gas Flow (Nominal) ACFM	103,000	1x99,000 2x79,000	2x35,000	3x44,000	110,000	2x67,000	89,000	55,000
Temperature °F	430-570	450-540	540	412	450-550	460	430	400
Fly Ash gr/DCF	1.8	0.5-6.5	1.6	1.2	0.06	1.6	3.2	2.4
HCl								
Average ppm	500	550	450	600	550	500	750	650
Maximum ppm	700	950	1800	900	950		1100	900
SO_2								
Average ppm	150	150	150	100	100	200	--	50
Maximum ppm	300	350	800		125		150	100
HF-Maximum ppm	15	25	20	15	15	10	15	5
Outlet Flue Gas from SDA								
Reference Conditions	11% O_2 dry	11% O_2 dry	11% O_2 dry	7% CO_2 wet	11% O_2 dry	11% O_2 dry		
Temperature °F	240	285	360	285	270	300	285	250
HCl ppm	20	30	60	60	30	60	60	60
SO_2 ppm	175	70	70	--	50	35	100	--
HF ppm	5	5	5	--	2	5	2	--
Particulate gr/DCF	0.02	0.02	0.03	0.02	0.02	0.013	0.012	0.02
Heavy Metal Requirements	YES	YES	YES	NO	YES	YES	YES	NO
Retrofit	YES	2 YES, 1 NO	YES	YES	YES	YES	YES	NO
Precollector	NO	YES (2)	NO	NO	YES	YES	NO	NO
Dust Collector	ESP	ESP	ESP	ESP	ESP	Pulse-Jet Baghouse	Pulse-Jet Baghouse	ESP
Operating Mode	S.P.	REC.& S.P.	S.P.	S.P.	REC.	REC.	S.P.	S.P.
START-UP	3/86	4/86	6/86	3/87	3/87	6/87	10/88	7/88

Inlet temperatures range from 400-570°F, typically somewhat higher than North American applications. Spray dryer outlet temperatures are controlled to between 240 and (with one exception) 300°F. The exception is two combined MSW and industrial incinerators in Hamburg, Germany, where the design outlet temperature of 360°F due to periods of very high inlet HCl to SO_2 ratios. All but the last two installations have successfully undergone performance and compliance testing. Performance tests for the last two systems are scheduled for the spring of 1989.

The Leverkusen installations are typical of MSW SDA systems which employ an ESP as the main dust collector. The systems are designed to meet emissions requirements when operating in the single pass or recycle mode. The design SDA outlet temeprature during recycle is 258°F and 240°F during single pass operation. Performance test results have been reported in previous papers (REF. 25) and are summarized in Table II.

TABLE II. LEVERKUSEN LINES 2 & 3 - PERFORMANCE TEST SUMMARY

	Average Values @ 11% O_2 Dry	Removal %
HCl Inlet, ppm	640	
Outlet, ppm	24	96
SO_2 Inlet, ppm	115	
Outlet, ppm	27	77
HF Inlet, ppm	9.4	
Outlet, ppm	0.25	97
Total Particulate, gr/dscf		<0.001
Trace Metal Emissions mg/NM^3		
Cadmium	0.002	
Colbalt	<0.0015	
Vanadium	<0.0015	
Chromium	0.008	
Lead	0.043	
Copper	0.008	
Thallium	<0.0015	
Mercury	0.646	
Arsenic	0.002	
Nickel	0.002	

Greater than 95% HCl removal, 75% SO_2 removal and 97% HF removal were achieved at an equivalence ratio of less than 1.3 (moles of alkali per equivalents of acid gases at system inlet). The acid gas removal efficiencies, acid gas emissions and lime consumption were all better than guaranteed. The acid gas, particulate and trace metal emissions are all well below the permitted levels. The trace metal values shown are for the total of particulate and vaporous forms. Total mercury emissions (0.646 mg/NM^3) are more than 99% in the vapor phase. These emissions can be decreased substantially by decreasing the SDA outlet temperature.

Kassel is the first Niro SDA MSW system to start-up which employs a fabric filter as the main dust collector. The Kassel municipal waste incinerator is situated in the Southern part of Germany. The incinerator burns household waste and incorporates two lines each with a capacity of 250 tons per day. The existing ESP's are used as dust collectors. Outlet dust emissions from these pre-filters is 0.04 gr/dscf. The incinerator operates 6,000 full load hours per year. The FGC system consists of a spray dryer absorber and a Luhr pulse-jet fabric filter with horizontal bags. The system is designed as a recycle system. Recycle slurry and slaked lime milk are mixed in a fast response head tank just before the atomizer. This provides optimal control such that the required half-hour outlet emission limits can be maintained even with fluctuations of HCl and SO_2 at the inlet to the FGC system. Performance test results are shown in Tables III and IV.

TABLE III. KASSEL - PERFORMANCE TEST RESULTS

	Inlet	Outlet	Average Removal %
Gas Flow, acfm	65,000		
Gas Temperature, °F	450	280	
HCl, ppm average	1,070	10	99+
range	600-1525	6-12	
SO_2, ppm average	125	10	90+
range	70-140	5-15	
Particulate, gr/scf @ 11% O2		0.0007	
range		<0.0001-0.0021	
Equivalence Ratio	less than 1.4		

Greater than 99% HCl and 90% SO_2 removals were measured with outlet emissions averaging 10 ppm each. The particulate emissions were well below the guarantee value of 0.013 grains per dry standard cubic foot. The equivalence ratio was below the 1.4 value guaranteed for 98.5% HCL and 82% SO_2 removal.

TABLE IV. KASSEL TRACE METAL EMISSIONS

Trace Metal Emissions mg/NM^3(dry)		Limits mg/NM^3
Class I		
. Cadmium	0.00027	
. Thallium	0	
. Mercury	0.00025	
	0.00052	<0.2
Class II		
. Arsenic	0.00225	
. Nickel	0.0032	
. Cobalt	0	
	0.00554	<1
Class III		
. Vanadium	0	
. Chromium	0.00123	
. Lead	0.00206	
. Copper	0.00192	
. Maganese	0	
	0.00521	<5

Measured particulate trace metal emissions are also very low for this system. The total emissions for all of the trace metals measured is about 0.011 mg/NM^3 (less than 5 x 10^{-6} gr/dscf) and are more than two orders of magnitude below the permitted levels.

The last two Niro MSW SDA systems to start-up in Europe are on the Carriere sur Seine and Amagerforbraendingen incinerators. Both of these systems are equipped with a new design roof gas disperser which has been designed for low abrasion. The two systems also incorporate a dual liquid distributor for separate introduction of lime slurry and dilution water into the atomizer wheel. This design allows for even faster response to changes in flue gas acid gas concentrations and simplifies the slurry feed system controls. These new design features are currently being evaluated for incorporation into our standard designs.

The FGC system at Carriere sur Seine started-up without problems in October 1988. The FGC system at Amagerforbraendingen in Copenhagen is a Volund/NKK system with Niro Atomizer and Flakt Danmark as sub-suppliers for the absorber/atomizer and pulse-jet filter respectively. This system was started in the fall of 1988, and has been running smoothly since then. Performance tests for these units are scheduled for spring 1989.

NORTH AMERICAN APPLICATIONS

To date, 17 MSW incinerators have committed to use Joy/Niro SDA systems to control acid gas and particulate emissions. All of these systems are grass-roots plants. Table V presents design conditions for the first seven units now under construction or in operation.

TABLE V. U.S. DESIGN CONDITIONS

TYPE WASTE		RDF	RDF	MSW
PROJECT		SEMASS Rochester, Massachussetts	West Palm Beach, Florida	York County, Pennsylvania
Inlet Flue Gas Conditions				
Flue Gas Flow (Nominal)	ACFM	2 x 212,000	2 x 197,000	3 x 95,000
Temperature	°F	350	350	350
Fly Ash	gr/SCF	10.0	7.2	5.5
HCl				
Average	ppm	400	710	500
Maximum	ppm	1000	1290	1000
SO_2				
Average	ppm	110	240	150
Maximum	ppm	330	430	300
HF				
Maximum	ppm	—	40	10
Outlet Flue Gas Conditions				
Reference Conditions		12% CO_2 dry	12% CO_2 dry	7% CO_2 dry
Temperature	°F	265	255	250
HCl	ppm	50 or	—	50 or
	% removal	90	90	90
SO_2	ppm	100 or	—	30 or
	% removal	65	70	70
HF	ppm	—	—	<1
	% removal	—	93	90
Particulate	gr/SCF	0.03	0.015	0.015
Dust Collector		Electrostatic Precipitator	Electrostatic Precipitator	Fabric Filter Pulseflo
START-UP		Spring 1988	Spring 1989	Winter 1990

These systems tend to be larger than European applications and are designed to operate with lower inlet and outlet flue gas temperatures. Design acid gas concentrations and removal efficiencies are similar to Europe. The first four systems are equipped with ESP's as the dust collectors while the remaining 13 systems will employ Joy PULSEFLO® fabric filters. All of these systems are designed for single pass operation, and incorporate fast response head tanks with lime addition controlled automatically based on acid gas continuous analyzers. The SEMASS plant has started-up and the West Palm Beach plant is in the initial start-up phase.

The SEMASS waste-to-energy plant located in southeastern Massachusetts was developed, by Energy Answers Corporation, as a regional waste processing facility. The plant is sized to incinerate up to 1,800 tons per day of MSW in two incinerators using a shred and burn technology. The incinerators are designed for semi-suspension firing. Process refined fuel (PRF) is blown onto the front waterwall by five fuel distributors. The light fraction burns in suspension while the heavy fraction falls to and burns on the grate. This type of firing leads to a high grain loading at the inlet of the spray dryer (up to 10 gr/scf). Overfire air is proportioned in three zones to aid in turbulence and provide oxygen for combustion in the furnace zone. From the furnace zone, the hot flue gas and entrained particulate passes up through the superheater, evaporator section, economizer and out through two tubular air preheaters. A bypass around the air preheater is included to allow for controlling the flue gas temperature leaving the boiler and entering the air quality control system.

The design SDA inlet temperature is 350°F although inlet temperatures down to 320°F were considered in the SDA system design. It is planned to test the SDA system at these lower inlet temperatures and determine optimum operating conditions to minimize overall plant operating costs, considering plant efficiency, atomizer power consumption and lime usage.

The SEMASS SDA systems employ one 41 foot diameter spray dryer absorber and a five field rigid frame electrode design ESP per incinerator train. There is a common lime slaking syetm for the two SDA's. The ESP has a nominal specific collection area of 650 and is designed to achieve the required emissions with one field out of service. Additional details of the SEMASS plant have been previously presented (REF. 7). The SEMASS incinerators started-up in July 1988. Initial opertion of the flue gas cleaning system was with the ESP's in service only. The spray dryers were put in service in late September. Operation to date has resulted in a clear stack with or without the spray dryer in service. Process optimization and fine tuning took place in the fall. The SDA systems have operated nearly continuously at outlet temperatures of 260-265°F with no drying problems.

The SEMASS SDA systems have recently completed an extensive compliance and performance program. The month long test program included emissions testing for each unit as well as removal efficiency tests and a determination of process consumables such as lime, power consumption, pressure drop and water consumption. A list of the procedures used and the pollutants measured are presented in Table VI on the following page. Results from these tests are expected to be available in May and will be reported at the earliest opportunity.

TABLE VI. SEMASS COMPLIANCE/PERFORMANCE TESTS

Parameter	Location	Test Method
Gas velocity	SDA inlet	EPA Method 1 & 2
Flow rate	ESP outlet	EPA Method 1 & 2
Gas composition for CO_2, O_2 and molecular weight	SDA inlet ESP outlet	EPA Method 3 EPA Method 3
Moisture content in stack gas	SDA inlet ESP outlet	EPA Method 4 EPA Method 4
Particulate emissions	SDA inlet ESP outlet	EPA Method 5d or 17 EPA Method 5d or 17
NO_x	SDA inlet ESP outlet	EPA Method 7e EPA Method 7e
Sulfur dioxide	SDA inlet	EPA Method 6c or 8
Emissions	ESP outlet	EPA Method 6c or 8
Sulfur trioxide	SDA inlet ESP outlet	EPA Method 8 EPA Method 8
Hydrogen chloride	SDA inlet	EPA Method 6b
Emissions	ESP outlet	EPA Method 6b
Non-methane hydrocarbons	ESP outlet	EPA Method 18 & 25a
Carbon monoxide	ESP outlet	EPA Method 10
Fluorides	SDA inlet ESP outlet	EPA Method 13a EPA Method 13a
Vinyl chloride	ESP outlet	EPA Method 10b
Reduced sulfur compounds	SDA inlet ESP outlet	EPA Method 16a EPA Method 16a
Dioxins/Furans	ESP outlet	EPA Method 5 modified
Pressure drop (static pressure)	SDA inlet ESP outlet	EPA Method 2 EPA Method 2
Opacity	ESP outlet	Opacity meter or equivalent
Gas temperature	SDA inlet ESP outlet	EPA Method 2 EPA Method 2
Power consumption	SDA control room building	Power meter or equivalent
Lime slurry consumption	Slaking building	Direct measurement of tank level
Lime slurry concentration	Slaking building	% solids & available lime in slurry

The West Palm Beach facility, designed by Babcock & Wilcox, processes MSW to produce a refuse derived fuel (RDF) for firing in two 1,000 tons per day incinerators. Each incinerator is equipped with a single SDA and a four field rigid frame electrode ESP. As with SEMASS, there is a common lime slaking system for the two SDA's.

The remainder of the North American applications are for mass burn incinerators. These applications are in engineering or under construction at five different sites being developed by Westinghouse Corporation or Ogden Martin Projects, Inc. All of these applications incorporate Joy low pressure PULSEFLO® fabric filters for dust collection. The filters are designed to achieve outlet particultate emissions of 0.015 grains per dry standard cubic foot corrected. The fabric filters have either four or six compartments, off-line cleaning, and net filtering ratios of 4:1 or less. The bags are woven fiberglass with a teflon coating and range in length from 14 to 20 foot. The majority of these units are scheduled to start-up in the next year.

CONCLUSIONS

Joy and Niro have jointly developed and applied an effective dry scrubbing process (SDA) to many differently types of combustion sources including mass burn and RDF fired MSW incinerators in Europe and North America. More than 23 systems are successfully operating meeting a wide range of emissions requirements for acid gases, particulate matter and trace elements. The majority of the European applications have completed performance and compliance tests. The units which have been tested have met all performance guarantees and continue to provide reliable effective flue gas clean-up.

European SDA systems are, for the most part, retrofitted on existing mass burn incinerators and employ EPS's as the dust collectors, although, there are a few notable exceptions. Both recycle and single pass operating modes are used which leads to design SDA outlet temperatures ranging from 240-300°F.

North American applications on the contrary are only applied to grass-roots plants. These plants are generally larger than the European and are designed for lower SDA inlet and outlet temperature operation. The first two facilities to start-up in North America have ESP's as the dust collectors, however, the majority of the systems will employ fabric filters. All of the systems are designed for fast response to changes in inlet gas conditions and all operate in the single pass mode. The first two trains have successfully started-up at the SEMASS facility and have recently completed compliance and performance tests.

Joy and Niro believe that our system designs will continue to set the industry standard for MSW dry scrubber applications throughout the world.

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SOURCE SEGREGATION STRATEGIES EFFECTING THE CHEMICAL PROPERTIES OF SOLID WASTE UTILIZED AS COMMERCIAL FUEL SUPPLEMENTS

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ABSTRACT

Municipal solid waste is a non-homogenous composition of refuse characteristically varying in terms of moisture, particle size, Btu value, chemical content and hazard potential. The reliance on landfills for disposal is a major national dilemma and waste of a recognized resource for energy use and recovery.

By the year 2000, the solid waste volumes could prove staggering. The alternative solutions encompassing waste to energy systems are controlling pollution, however, not without controversy. Multi-media transfer of pollutants and disposal of contaminated ash magnify the issue. With emphasis on source segregation and recycling and a resurgence of economic incentives, solid waste resources are expected to evolve into saleable commercial fuels supplementing the energy requirements of various thermal conversion systems having applicability to municipal and industrial solid waste.

INTRODUCTION

Throughout this country as well as world-wide, industrial manufacturers and consumers are recognizing the economic burdens and environmental liabilities being created by solid waste. As a categorical waste stream, municipal solid waste (MSW) can be characterized as a heterogeneous mixture of combustible and non-combustible refuse varying in physical and chemical properties as well as hazard potential. The refuse materials comprising these solid waste mixtures can vary significantly. This variability is directly influencing the heating value as well as the residual by-product characteristics of waste-to-energy systems.

With the growing interest in waste-to-energy systems, emphasis has been placed on the system designs which can improve performance efficiencies and control residual pollutants. Industries and municipalities are reassessing existing waste management practices (many of which still rely on land disposal) in an effort to reduce: waste volumes, reliance on landfills and environmental liabilities as well as to identify opportunities providing economic gains. The focus on waste prevention and recycling will provide many generators with alternative outlets for resaleable refuse. The impetus for change will be driven by economics as well as regulatory mandates.

The volumes of waste which can be source segregated to enhance recycling of solid waste as commercial fuel supplements is staggering. The U.S. Environmental Protection Agency estimates that the quantities of solid waste will reach 380 billion pounds by the year 2000. Nearly 85% of the estimated solid waste quantities still rely on landfills for disposal. The landfillable material represents, to a great degree, wasted energy and resources for reuseable products having economic value. As concerns over landfill capacity and acceptance magnify in terms of cost, risk and waste pretreatment, industrial generators will turn to alternative technologies effective in converting waste to energy.

One of the problems associated with the variability in solid waste mixtures is the appreciable quantities of non-combustible refuse material negating both the heating value of the solid fuel and the reuse potential for the solid ash residuals. Many of the constituents comprising the solid fuels can be segregated enhancing both the heating value of solid waste mixtures as well as individual wastestreams. In this regard, industries can strategically develop blends of solid waste fuel for commercial resale and use.

Recognizing the on-going problems with land disposal, strategies are presented in this paper for source segregating and recycling various industrial types of solid waste as commercially saleable fuel supplements. With environmental compliance standards becoming increasingly stringent, emphasis is placed on the effects of source segregation on MSW characteristics with particular focus on categorical groups creating high levels of ash containing concentrated hazardous substances.

WASTESTREAMS COMPRISING MSW CHARACTERISTICS

In order to assess the impacts of source segregation and recycling on the characteristics and fuel value of solid waste, it is important that the composition and Btu value of the gross 'as-discarded' wastestreams be well understood along with the specific components comprising the waste. It was previously stated that MSW is a defined heterogeneous mixture of combustible and non-combustible refuse varying in heating value, chemical content and hazard potential. The amount of non-combustibles in MSW can be significant depending on the regions generating the waste. The non-combustibles will characteristically average 20 to 30% by weight.

The heating values of MSW characteristically average between 4000-5000 Btu/lb. The different types of MSW used as fuel in its 'as-discarded' form will vary with the various types of RDF. Table 1 provides a comparative summary of RDF fuel specifications and it is obvious that gross separation to remove ferrous metals, grit, glass and other inorganics coupled with shredding, to reduce fuel size, improves energy value while providing optimum control over the combustion process. The extent of source segregation will influence the economic value of recyclables as well as the resale value of alternative fuel sources.

Table 1. Summary of MSW Fuel Types

Parameter	RDF-1	RDF-2	RDF-3	RDF-4
Product Size	*	95% <4-4.5 in.	95% <1.5 in.	95% >10 mesh
Moisture Content	*	15-25%	12-20%	1-5%
Ash content	*	12-20%	15-20%	10-20%
Btu/lb	*	4500-5500	5000-6000	6500-7500
Density lb/ft^3	*	3-7	2-6	25-35
RDF Yield (RDF/Raw MSW)	1	70-80%	55-65%	

RDF1 - MSW used a Fuel in its 'as-discarded' form without processing (1)

EPA estimates that a third of the landfills in the country are expected to close in the next five years. Source segregation and materials recycling remains a very controversial and under utilized solid waste management strategy. Of the more than 300 billion pounds of solid waste produced, only about 10% is recycled annually. Future trends in solid waste generation suggest that the disposal dilemma will magnify. Table 2 provides a summary of categorical solid waste materials disposed in the municipal waste stream.

According to Franklin Associates, plastics and paper are the fastest growing segments of MSW. In 1987, the plastics industry produced a total of 55 billion pounds of plastics with the public throwing away approximately 38 billion pounds, accounting for 10% of the annual waste generation. More than half of the discarded plastics are found in the form of packaging. Packaging, about one third by weight of consumer waste, consists primarily of paper (48%), glass (27%) and plastics (11%). As a percentage of MSW, packaging and container residues show a declining trend in waste generation rates.

Table 2. Materials Disposed Into The Municipal Waste Stream
(In Millions of Tons and Percent) (2)

	1970		1984		2000	
Materials	Tons	%	Tons	%	Tons	%
Paper/paperboard	36.5	33.1	49.4	37.1	65.1	41.0
Glass	12.5	11.3	12.9	9.7	12.1	7.6
Metals	13.5	12.2	12.8	9.6	14.3	9.0
Plastics	3.0	2.7	9.6	7.2	15.5	9.8
Rubber & leather	3.0	2.7	3.3	2.5	3.8	2.4
Textiles	2.2	2.0	2.8	2.1	3.5	2.2
Wood	4.0	3.6	3.6	5.1	3.8	3.8
Other	----	0.1	0.1	0.1	0.1	0.1
Food Wastes	12.7	11.5	10.8	8.1	10.8	6.8
Yard Wastes	21.0	19.0	23.8	17.9	24.4	15.3
Miscellaneous Inorganic Wastes	1.8	1.6	2.4	1.8	3.1	2.0
TOTAL	110.3	100	133	100	158.8	100

Food and yard wastes as well as other inorganic residues comprise the category of miscellaneous wastes, identified on Table 3. As noted on Table 3, durable goods are increasing rather slowly but nondurables are growing rapidly.

Table 3. Products Discarded Into The Municipal Waste Stream* (3)
(In millions of tons and percent)

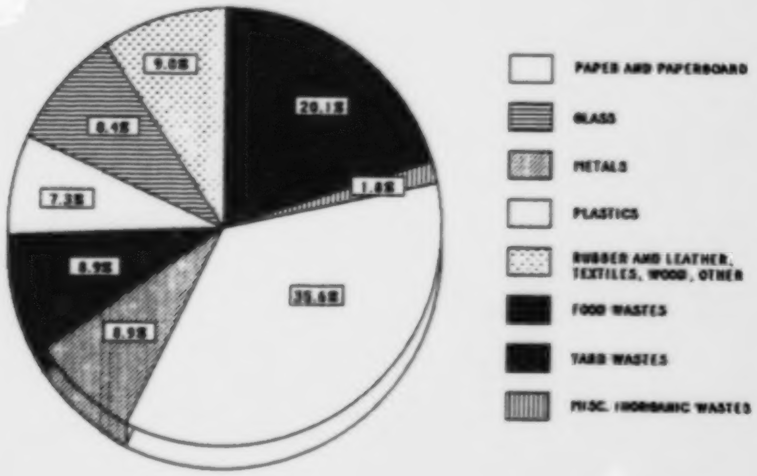
	1970		1986		2000	
Products	tons	%	tons	%	tons	%
Durable Goods	13.9	12.4	19.2	13.6	23.0	13.6
Nondurable Goods	21.4	19.0	35.4	25.1	47.5	29.1
Containers & Packaging	39.3	34.9	42.7	30.3	50.7	30.0
Other Wastes	37.8	33.6	43.4	30.8	47.5	28.1
	112.5	100.0	140.8	100.0	168.8	100.0

*Wastes discarded after materials recovery and before energy recovery.

Details may not add to totals due to rounding.[2]

According to EPA's estimates, discarded refuse after recovery of materials still contains appreciable percentages of recyclable fuel supplements. Figure 1 graphically illustrates the percent of materials discarded in MSW.

Figure 1. Materials discarded into the municipal waste stream in 1986, in percent of total (4).



Some of the specific trends influencing future MSW characteristics and fuel values are summarized as follows:

PAPER AND PAPERBOARD

By the year 2000, refuse comprised of paper and paperboard will be generated at an annual rate estimated at 66 million tons. The high energy value enhances the Btu content of heterogeneous refuse, however there are other applications which would also enhance industrial solid fuels in hazardous incineration operations.

WOOD

The cellulosic derivatives reclaimed from wood products can be formulated into alternative fuel supplements as well as fillers for residues requiring improved material handling specifications.

DURABLE GOODS

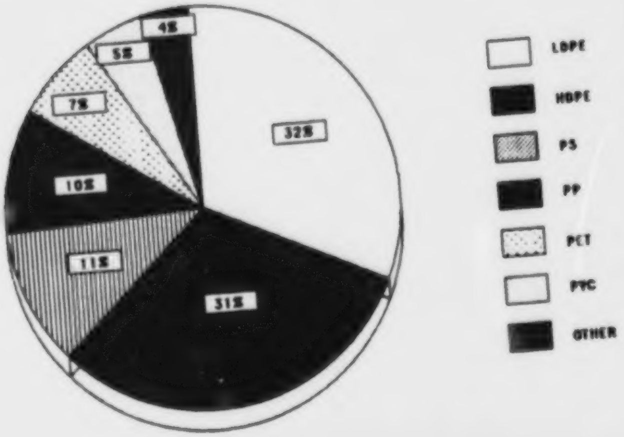
Rubber tires, diversified synthetic rubber products and textiles also provide unique characterisitcs in terms of supplementing alternative fuels in cement kilns, pyrolytic systems and high temperature incinerators.

PLASTICS

The increased volumes of refuse comprising this category are expected to steadily increase even though the weight percentages will not be significantly higher. Thermoplastics account for 87% (6) of the plastics sold and they present the most recyclable form of plastics with source segregation. The resins commonly used in consumer packaging applications are schematically illustrated on Figure 2. Typical sources are summarized as follow:

- o Low density polyethylene (LDPE): films, bags
- o High density polyethylene (HDPE): jugs, motor oil bottles, household containers
- o Polyethylene terephthalate (PET): beverage bottles
- o Polystyrene (PS): cups, trays, food containers
- o Polyvinyl chloride (PVC): construction, plumbing
- o Polypropylene (PP): diaper linings, battery cases

Figure 2. U.S. Total 1987 Production of Packaging Resins 12.7 Million Pounds (5)



The ultimate composition of MSW mixtures is summarized on Table 4.0

Table 4.0 Analysis of MSW and Coarse RDF Used in Direct Firing (7)

Parameter	MSW wt%	Coarse RDF wt %
Moisture	25.2	26.4
Carbon	25.6	26.8
Hydrogen	3.4	3.6
Oxygen	20.3	21.3
Nitrogen	0.5	0.5
Chlorine	0.45	0.48
Sulfur	0.15	0.17
Inorganics	24.4	20.7
TOTAL	100.0	100.0
Proximate Analysis		
Moisture	25.2	26.4
Ash	24.9	21.1
Combustibles (fixed carbon & volatile matter)	49.9	52.5

The ash content and heating value of the MSW mixtures varies as a function of changes in composition. Table 5.0 provides a typical profile of reference values (dry wt. basis) corresponding to individual wastestreams comprising the ultimate MSW composition.

Table 5.0 Wastestream Characteristics

	Processed Fuel (Dry wt. basis) (9)		
	Ash	Btu/lb	% Fixed Carbon
Newsprint	1.40	8560	12.20
Paper Products	5.15	8120	9.50
Textiles	5.10	8676	11.10
Plastics (film)	12.40	15,510	0.07
Plastics (rigid)	6.38	16,824	4.77
Food waste	7.51	8283	12.80
Wood	4.09	8932	15.10
Yard Waste	18.30	7442	15.10
Sweepings	33.50	6107	6.65

Typical moisture (dry) 25.2%
Ferrous, aluminum, glass, nonferrous no Btu value

The trends in recycling of paper products and plastics can reduce the heating value of MSW blends. The losses generally have to be off-set by segregating and/or reducing yard wastes and/or removing metallic-bearing residues.

Source segregating inorganic residues and metallics can reduce many of the problems encountered with bottom ash. Certain metals, i.e. lead, cadmium, and mercury will volatize with combustion in addition to contributing to ash properties.

METALS IN ASH - POSSIBLE SOURCES AND SEGREGATION SCHEMES

MSW ash continues to undergo regulatory scrutiny relative to environmental risks posed by land disposal or potential reuse as aggregate. The resulting ash and its potentially hazardous characteristics are critical factors in the development and use of a commercial fuel supplement.

As referenced on Table 6.0, the concentrations of some of the major inorganic constituents typically detected in fly ash, combined ash and bottom ash varies significantly. The residual ash composition provides unique properties as a pozzolana material. In combination with alkaline based reagents, these pozzolanas can provide cementitious properties and, in some cases, a degree of chemical immobilization. However, with upcoming changes in hazardous waste definitions and the stringent enforcement of land disposal restrictions, refuse strategies for ash contaminated with metals and organics will continue to be scrutinized. Preventing and/or reducing the loadings of residual pollutants in ash requires a close examination of source segregation and control strategies. The solid waste fuel composition will be dictated by commercial use and by-product characteristics.

One of the strategies focuses on the separation of ferrous metals from ash. The West German process, the Thyssen Henschel vertical Grinder Mill (9), involves magnets, grinding, dedusting and sorting of ash according to content, size and weight. With the ash from mass-burn incinerators, the system is capable of grinding the ash to free ferrous material and breaking down silicate (glass). The iron is reclaimed with magnets and wind-sifts the salvage by stair-stepping it down a 'z-box' while simultaneously blowing air upwards against it. The removed fly ash is baghoused and dumped while larger particles are sorted by size in a trammel. The end product is a dense metal product that has been cleaned of paper, dust and other contaminants and is 99%$^+$ ferrous. From an environmental and disposal perspective the grinder reduces solid waste, which has already been knocked down 4 to 1 by incineration; another 75 percent. Refuse contributing to the loadings of total and leachable metals has been significantly minimized. In addition, the reclaimed ferrous material has a resale value of $75 - $125/ton, making the system attractive for implementation.

Waste tires continue to be major disposal problems and they can also contribute to residual ash containing hazardous constituents. The industrial residues generated in the tire manufacturing sector display some of the same characteristics as waste tires, however the total loadings of metals and contaminated oils could prove more hazardous. As referenced on Table 7, a typical tire char ash will vary in composition as a function of the feed.

Table 6. Ranges of Concentrations of Some of the Major Inorganic Constituents in Fly Ash, Combined Ash, and Bottom Ash From Municipal Waste Incinerators in ug/g (ppm) (10)

Parameter	Fly Ash	Combined Bottom and Fly Ash	Bottom Ash
Arsenic	15 - 750	2.9-50	1.3-24.6
Barium	88 -9,000	79-2,7000	47-2,000
Cadmium	<5-2,210	0.18-100	1.1-46
Chromium	21-1,900	12-1,500	13-520
Lead	200-26,600	31-36,600	110-5,000
Mercury	0.9-35	0.05-17.5	ND-1.9
Selenium	0.48-15.6	0.10-50	ND-2.5
Aluminum	5,300-176,000	5,000-60,000	ND-38
Calcium	13,960-270,000	4,100-85,000	5,900-69,500
Iron	900-87,000	690-133,500	1,000-133,500
Magnesium	2,150-21,000	700-16,000	880-10,100
Nickel	9.9-1,996	13-12,910	9-226
Silicon	1,783-266,000		1,333-188,300
Sodium	9,780-49,500	1,100-33,300	1,800-33,300
Tin	300-12,500	13-380	40-800
Zinc	2,800-152,000	92-46,000	200-12,400

Source segregation, shredding and grinding systems can minimize the impacts of zinc and other pollutants, producing a fuel blend that provides higher Btu values.

Table 7. Average Chemical Analysis of a Typical Tire Char Ash (11)

Element (wt. %)	Non-Belted (Ash 15.4 wt. %)	Steel-Belted (Ash 15.0 wt %)	Fiberglass-Belted (Ash 13.7 wt. %)
Antimony	<0.03	<0.03	<0.03
Aluminum	2.40	1.50	2.40
Chromium	0.15	0.13	0.16
Iron	2.00	1.10	2.00
Lead	0.16	0.18	0.25
Barium	0.27	0.25	0.60
Magnesium	2.30	1.50	2.50
Phosphorous	1.60	0.80	1.40
Potassium	0.38	0.32	0.36
SiO_2	19.62	19.09	21.31
ZnO	45.45	52.33	45.64
TiO_2	11.99	14.27	10.28
CaO	6.04	5.06	6.08
Sulfur	2.67	2.77	1.27

Source segregation of solid industrial waste followed by strategic separation of waste components can have a major effect on the overall loadings of pollutants in ash. In many cases the separation of components will be dictated by economic feasability as well as the effectiveness of source collection and segregation policies.

Some sources of heavy metals which warrant segregation are:

- o Cadmium: industrial fungicides, photographic wastes, batteries, ceramics, plastics, rubber products
- o Chromium: automobile parts, paints, glass, photographic residues, textile residues
- o Lead: storage batteries, nonferrous solid residues, leaded glass, paints
- o Nickel: foundry residues, steel products, motor vehicle parts, printing residues
- o Zinc: yarn/fibers, newsprint production residues, rubber residues, tires, paints
- o Mercury: batteries, pesticides, paints, plastics, machinery sealants, film production residues

Industries and municipalities must keep in mind that the majority of the metals found in the MSW are contained in inert items which can be removed by segregation and component separation. Nearly sixty percent, if not more, of the cadmium and mercury in solid wastes are usually associated with nicad, alkaline, and mercury batteries. Removing these, along with solder-bearing items such as electrical equipment, plated ferrous metals, lighting fixtures can significantly reduce the amount of lead, cadmium, tin, mercury and zinc in the solid fuels prior to thermal treatment. Besides the improvement in ash characteristics, the metal separation schemes will significantly reduce metal pollutants in flue gas emission. Overall, source separation of metallics will improve combustion efficiencies and disposal rates.

HEATING VALUES ASSOCIATED WITH INDIVIDUAL WASTE CATEGORIES AND COMPONENTS

This section discusses some of the energy values associated with individual wastestreams and components comprising MSW fuels. The subject matter focuses primarily on the heating values of industrial wastestreams amenable to source segregation and component separation.

POLYMERS AND PLASTICS

Mylar film, lucite, nylon-6, phenol-formaldehyde resin, polybutadiene, polyvinylidene chloride, polyethylene, polystyrene, polypropylene, polyvinylchloride and teflon are among the vast array of polymer and plastic solids comprising solid wastes in the plastics and packaging categories. As previously stated, the plastics and polymeric resins provide an excellent fuel source and they enhance the energy value of MSW massburn and RDF mixtures. The individual components characteristically display energy values ranging from (12):

8,000 to 19,900 Btu/lb for non-halogenated plastic chips, and
2,000 to 8,000 Btu/lb for halogenated organic waste types

Component separation schemes are allowing many of the plastic wastes to be purified for product reclamation and reuse. Separation technologies are generating clean, well-separated (99.9%) granulated plastic chips. With the development of end use markets. PET and HDPE waste containers will prove economically attractive for reclamation. If the desired purity and separation of aluminum caps and labels cannot be attained, the waste materials can be segregated, granulated and used as high energy commercial fuel supplements in hazardous incineration systems.

RUBBER PRODUCTS (13)

There are four types of rubber used in tires; natural, styrene-butadiene, polyisoprene and polybutadiene. A typical compounding formula for passenger tire treads made out of natural rubber provides excellent properties as a high energy fuel. Destruction distillation of tires can yield solid, liquid and combustible gas fuel sources. For the solid categories of discarded tires, the energy value can range from 13,000 to 17,000 Btu/lb. Shredding and component separation systems can enhance the energy value while reducing ash and metallic loadings. New technologies are allowing for adjustment of particle size and the separation of rubber, steel zinc oxide and fiber components. The generation of a rubber dust or a pyrolytic solid residue can provide excellent heating values as well as the impetus for material reuse.

PAPER (14)

With variety of waste sources, individual wastestreams can be segregated and shredded yielding the following characteristics:

Brown Paper

As received, typical energy value is 7260 Btu/lb yielding an ash approximated at 1%.

Corrugated Box

Waste solids typically displaying energy values estimated at 7047 Btu/lb yielding ash contents approximated at 5.0%

Newsprint

Waste solids at room temperature typically display energy values estimated at 7980 Btu/lb yeilding ash contents approximated at 1.4%.

Miscellaneous

Plastic coated papers, waxed milk cartons and magazines can be blended yielding energy values ranging from 5250 Btu/lb to 11,300 Btu/lb.

Miscellaneous Solids (15)

Individual wastestreams categorized as combustible and non-combustible solid wastes display variable energy values, some of which are summarized as follows:.

tree materials	700-900 Btu/lb
artificial graphite	12,000-13,000 Btu/lb
leather	7,000-8,000 Btu/lb
paraffin wax	19,000-20,000 Btu/lb
food products	1,000-17,000 Btu/lb
agricultural residues	7,000-8,500 Btu/lb
plant materials	6,000-7,500 Btu/lb

UTILIZING SEGREGATED SOURCES OF SOLID WASTE AS COMMERCIAL FUEL SUPPLEMENTS

Solid waste management strategies encompassing source segregation and spearation are resulting in a number of innovative technologies for cleaning and reusing solid waste as commercial fuel supplements. In addition to the multitude of waste to energy systems, there are a number of industrial thermal conversion and destruction systems relying on commercial fuel supplements as alternative energy sources. The utilization of these fuels will improve its marketability in both the industrial solid as well as hazardous thermal incineration areas. As landfill capacities decrease, there will undoubtedly be additional requirements for thermal destruction and recycling. With emphasis on waste prevention and recycling, hazardous waste thermal units will find that solvent blends, used as liquid fuel supplements, will decrease in terms of their availability. The anticipated declines in their rates of generation will dictate the use of alternative fuel sources especially with MSW blends and industrial residue displaying low energy values. The alternative fuel supplements will also need to compensate for losses in the heating value of combined MSW residuals if plastic, paper and rubber product recycling proves economically attractive.

Taking into account recent developmental activities focusing on waste utilization, various technical strategies are identified and discussed in terms of resource utilization.

UTILIZING ASPHALT ROOFING WASTES

With recognized increases in disposal costs, the option of converting asphaltic-based solid residues into fuel supplements is gaining interest. Total asphalt roofing products' waste has been estimated to represent about 7.3×10^{13} Btu annually (16). Various alternative fuel utilization strategies have been funded by the Manville Service Corporation. Segregating this stream will enhance fuel value and the economic gains associated with material reuse.

PLASTIC WASTES RECOVERY SYSTEM

The range of compatability of thermoplastics with other plastics is relatively small, necessitating separation systems for recycled plastics on the basis of polymer types, viscosity ranges, types of additives and hazards associated with thermally treated by-products. The mechanisms for separating and pretreating plastics has been coupled with design reactors capable of converting plastics into fuel oil. In 1985, Procedyne Corporation (17) successfully converted polyethylene terephthalate and polyvinyl chloride into fuel oil.

In 1986, an estimated 2.9 billion pounds of plastics were used in plastic bottles. Nearly 62% of this total is comprised of high density polyethylene (18). In comparison to the recycling of paper and aluminum, plastic recycling is expected to have a major impact providing source segregation and separation systems are adequately utilized. The Plastics Recycling Foundation is projecting market demands for reclamation with:

- o PET recyclable solids growing from 150 million ponds to 600 million pounds per year by the mid-1990's and,
- o HDPE recyclable solids increasing from 50 million pounds (1987) to more than 500 million pounds per year by the mid-1990's

Besides fuel alternatives, the reclaimed plastics are finding wide applicability to the production of plastic lumber, boat docks, landscaping ties, outdoor furniture, car stoppers and farm structures. Additional markets including specialized fuel alternatives will develop, however; source separation is critical to future success. Markets are also evolving for commingled plastics which cannot be segregated.

AUTO SHREDDER FUEL

Energroup, Inc. has evaluated and demonstrated the feasibility of using automobile shredder residue (fluff) as a fuel for in-plant power consumption. Energy values averaged 5000 Btu/lb for a number of sources of shredded fluff (19). The work efforts identified this waste category as a viable energy resource; economically preferential in comparison to wasting it by landfills.

WASTE TIRES

Only 20% of the over 200 million tires discarded annually nationwide are currently utilized as a resource for energy recovery. Alternative fuel (20) uses offer energy-conservation opportunities through their use as:

- o solid fuel (displaced energy equivalent to 15,000 Btu/lb)
- o virgin rubber compound in traditional rubber products (34,000 to 40,000 Btu/lb)
- o asphalt additive for paving applications (90,000 Btu/lb)

The energy estimates include both the energy of the displaced fuel and material and the energy consumed preparing the tires for the cited uses.

Eastern Products Corp. has demonstrated that tire-derived fuel can be co-fired with oil in a concentration of up to 16% of the Btu input to the boiler. Combustion efficiency was approximately the same as when burning out only.

CELLULOSIC WASTE

Numerous studies have been funded by DOE assessing the feasibility of converting cellulosic wastes into liquid hydrocarbon fuels. A staged reactor was defined to convert biomass to useful oils. Process stages of the operation suggest the feasibility of reclaiming a fuel oil for commercial use.

SUMMARY

Throughout the country, centralized processing systems referred to as resource recovery and/or waste-to energy systems are being developed and utilized as the solution to the land disposal dilemma and rising disposal costs. The MSW can be collected and separated yielding various types of solid fuels, each displaying variable characteristics relative to particle size, density, heating value, residual ash and hazard potential. The overall strategy can be viewed as a control system that is and will continue to be faced with regulatory and environmental issues; some of which are the same as those the solution is supposed to replace.

EPA recently outlined its 'Agenda for Action', which refocuses solid waste management strategies on solid waste recycling and reuse. The goals are clearly focused on the reduced reliance on landfills. Industrial generators of solid waste containing hazardous substances may soon be in the 'arena' of hazardous waste. Upcoming changes in hazardous waste definitions, as well as land disposal acceptance criteria for hazardous and as 'specialty' solid wastes will require more emphasis on source reduction and thermal destruction. Waste minimization efforts are, in many cases, proving successful in replacing combustible liquids with aqueous-based substitutes. The benefits are recognized by the elimination/reduction of the risks posed by the wastes, however, in some respects they are creating additional gaps in available fuel supplements. Fluidized bed, liquid injection, cement kilns and rotary kilns are the thermal destruction systems relying on fuel supplements comprised of industrial waste. The fuel demands and the anticipated changes in thermal destruction performance standards for industrial and municipal solid waste will provide the impetus to reassess source segregation and separation techniques; not just for product recycling, but also for the creation of solid fuel supplements displaying high energy values and resale potential.

Source segregation and component separation strategies are available alternatives to central processing facilities, yet it is conceivable that the resulting by-products of source treatment will supplement the energy demands of these central thermal conversion systems. The source segregation and separation strategies must account for the end use markets. In terms of generating commercial fuel supplements, the generator must project end-use specifications that will dictate the degree of segregation and component separation. There is already a considerable interest in developing categorical groups of solid waste as commercial fuels, however, in order to make them economically attractive, the fuel specifications must be well understood and market must be well defined.

The disposal trends into the year 2000 clearly substantiate the potential for resource utilization. Many of the solid waste categories providing heating values with combined solid refuse will be candidates for recycling. With anticipated escalation in the recycling of paper products, yard wastes, tires and wood products, industries can anticipate (21):

- improved combustion efficiencies
- reduced pollutant loadings in the multi-media by-products of thermal conversion systems
- increased disposal rates
- reduced operating and disposal costs
- better conditions for complying with existing as well as future thermal destruction standards
- seasonal variations in refuse moisture content, however, the impacts could be minimized with the recycling of wood and yard wastes
- future demands for fuel supplements providing high energy values and low ash

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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PROCESSING HOUSEHOLD RECYCLABLES

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International Conference of Municipal Waste Combustion
Hollywood, Fl
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Lessons from a Stone-Age Recycling Camp

I started out in recycling as volunteer general manager of the once-a-month drop-off center of the Coalition on Environment, Pelham, New York. That was in May, 1971, the spring next after Earth Day, April 22, 1970. I obtained permission to use the municipally-owned site, I wrote and placed press releases, I recruited volunteer workers, I rented the roll-off boxes, I arranged the sale of the scrap and I turned the checks over to the treasurer. Monthly on the appointed weekend, townspeople dutifully brought us their bottles, cans and used newspaper in their 7-mile-per-gallon station wagons.

After we had our stone-age recycling camp running for a year and I was able to delegate some of the tasks, I asked the question that was most dangerous to my ego and to those of my fellow volunteers: Were we doing any good at all? With some trepidation we compared the sum of our weight tickets for scrap sold, with the reported tonnage of collected solid waste produced in our town and weighed out at the neighboring transfer station. Result: on our best month (usually May or June), we diverted around 0.3 percent of the town's municipal solid waste to recycling. We had fun together, we felt good, we spread the good word about the environment, we engaged in a monthly act of worship--but we clearly were not the solution to the garbage problem.

I was a slow learner. It took me four years of that weekend volunteer labor to realize three lessons:

(1) You can't recycle what you can't sell.

(2) Recycling, if it is ever to make the big time as part of the solution to the solid waste problem, must be industrialized, and institutionalized.

(3) To be successful, recycling must be convenient for the participants.

Those who collect materials for recycling and fail to sell them leave heaps. I am pleased to say that our drop-off center sold what it took in and left nothing behind. So we were successful on the first point, but a failure on the second and third--we didn't make the big time, and it sure wasn't convenient.

Measures of Performance

The fundamental measure of performance for source separation programs is their diversion rate: mass of material diverted to scrap markets, divided by total mass of materials discarded.

The next most important quantitative fact is the participation rate--it is 100 percent if every household in the target area is setting out its recyclables as instructed.

Factors Contributing to Performance

There is widespread, although not universal, agreement that participation rate is dependent upon convenience factors such as:

- Availability of curbside collection service for recyclables, as opposed to my stone-age example of requiring that all the materials be delivered by participants to a drop-off site;
- Minimum number of separate categories in the set-out, that is, as many materials as feasible to be collected in one container.

Now admittedly it would be most convenient if there were no need at all for separation of materials at the source. Indeed, over the past 20 years many attempts have been made to get clean scrap from dirty garbage, and simultaneously prepare refuse-derived fuel or material suitable for sale after composting. Each one has succeeded technically to some degree. Only one or two have lasted, however, and in the long-lasting cases the only material recovered for its scrap value, and actually sold on any regular basis, has been iron. Capital and operating costs have more than consumed the sum of materials and energy revenues, and net tipping fees exceed those for mass burning with energy recovery in the same vicinity. Despite these hard facts, getting clean scrap from dirty garbage remains a kind of Holy Grail for technicians in the solid waste business.

Until the Grail is found, however, we have decided that the best adjustment to reality is to encourage integrated programs: source separation at the household and the business for the recovery of recyclable scrap, separate leaf and yard waste collections designed to produce a salable compost, sturdy, well-operated mass burners for energy production from the non-recyclable remainder, and the inevitable landfill for ash and non-processables.

Source separation is the first step in getting clean scrap from clean trash, uncontaminated with moisture, fats, greases, vacuum cleaner catch, cat litter, used disposable diapers and so forth. What kind of source separation works best? Since so much experimentation and pioneering work has been going on for the past two decades, and continues now, no one can honestly pick out a single best method. What is best for the inner city (buyback centers? dropoffs?) may not be best for less densely settled areas (separate curbside collection?).

Regarding the number of categories into which participants must divide their materials to make them acceptable, higher participation rates seem to correlate loosely with fewer categories, because fewer categories means greater convenience. A survey by Zogby Associates for the engineering firm of William F. Cosulich in upstate New York two years ago found respondents very positive about a two-container system, one dirty (non-recyclable), one clean, in which all recyclables were accumulated in a single can. The percentage expressing willingness to cooperate was reduced somewhat for a three-container system. At four cans, the willing percentage dropped off steeply.

Whether these results carry over into actual experience is not yet certain; there are too few two-can systems in existence yet. Certainly the existing programs in New England involving three categories, dirty, used newspaper, and commingled container materials, have been popular and have reached participation rates consistently better than 50 percent.

Conversely we have observed some inconvenient programs: seven different cans in the kitchen and at the curbside has led to political rebellion in some towns in New Jersey. 1987 brought us the first instance of a citizen activist group forming against recycling, in the Borough of Lincoln Park, Morris County. Recyclables were required to be, first, divided into multiple containers, then delivered to the town drop-off center. Recyclables found by trash collectors in the regular garbage collection made the offenders liable for enforcement action. Residents objected that having paid their taxes for contract trash hauling, they should not be subject to the additional burden of carrying a portion of their discards to the recycling drop-off.

Intermediate Processing

The fewer the categories of recyclables in the set-outs, the more convenient for the the participants. However, a greater degree of commingling poses a technical challenge for the receiver of the collections. To be effective, the sorting and preparation of the scrap must be industrialized. Over the past 15 years a new type of scrap materials processing industry has been developing: the intermediate processing facility (IPF), also known as materials recovery facility (MRF).

Contrasted with Traditional Scrap Business: Recycling Economics

Intermediate processing is different from traditional scrap businesses because it handles a mixture of materials, largely from household discards, all of low value per ton compared to traditional scrap.

Traditional scrap businesses have focussed on commercial and industrial scrap, that typical comes in the gate of the processing facility as a single material--scrap iron, scrap copper tubing, etc.--and has a high value per ton based upon its relative purity. By contrast, traditional scrap dealers have not been enthusiastic about any sort of post-consumer scrap except aluminum cans. A traditional scrap business relies upon a collection network that responds to the price being paid at the scrap dealer's door. When user-industry demand is high for, say, scrap iron from old radiators, the dealer passes the demand through in the form of the price paid at his door, and the material flows. When the dealer cannot sell it, he reduces the price he pays his collectors, and the flow diminishes or stops. As a result, the price is generally kept from going down to zero.

But household post-consumer waste cannot be modulated in such a neat fashion. It is generated regardless of prices paid in the scrap market. To cite a painful and timely example, for the past 4 months used newspaper in the Northeast has fetched a zero or even a negative price. For example, Garden State Paper Company, which pioneered the deinking of used news in 1962, and has been a major market for that material ever since, has recently charged some suppliers a tipping fee to take used newspaper. To a degree, the paper can be warehoused for future use, but if the supply glut persists, the excess paper will have to be burned or buried, lest the accumulators of the material leave heaps.

In summary, the pricing of intermediate processing services must take account of the fact that the materials supply, being primarily household discards, cannot be turned off in a short time. The rate at which we buy, read, and discard newspapers is not sensitive to the value of the used newspaper in the scrap paper market. Supply will at times exceed demand for some or all materials, consequently a negative price at the gate, otherwise known as a tipping fee, should be anticipated.

Inside an Intermediate Processing Facility

In the facility, a combination of mechanical sorting equipment and human labor is employed to separate commingled recyclables from one another, then to size-reduce them and otherwise prepare them for sufficiently dense and economical shipment.

Mechanical sorting equipment is typically adapted from pre-existing industries, for example:

- the milling of metallic ores, in which separation is accomplished by relative density or magnetic properties;
- the preparation of wood products for pulping and papermaking, and the upgrading of wood wastes for use as fuels on the paper mill site;
- sorting by size and density in traditional scrap metals processing;
- crushing and screening of stone for use as aggregate in concretes;
- removal of moulding sand from castings in foundries.

If a totally commingled approach is attempted, that is, scrap papers mixed with bottles, cans and plastic jugs in the incoming stream, the initial challenge is the selection of a primary sorting device that produces a distinct division between the paper products and the container materials, leaving the glass essentially unbroken.

Glass, typically worth $40 to $60 per ton, currently must be sorted for color by hand. Automatic color sorters have been demonstrated, but their cost-effectiveness still has not surpassed that of human beings in a comfortable work setting. Plastics and some paper products require hand-sorting as well.

The attached flow chart will give a sense of the way in which mechanical and human effort are combined.

The Working Environment

People are the most expensive unit process in the intermediate processing facility. Therefore not only do they require first consideration in the arrangement of the working environment, but also they must be assigned in a rational way in order to yield the greatest value added for the cost of their labor.

Those of you who may have visited some of the existing examples of the intermediate processing genre have probably been struck by the noise, dust, absence of temperature controls, and lack of adequate lighting.

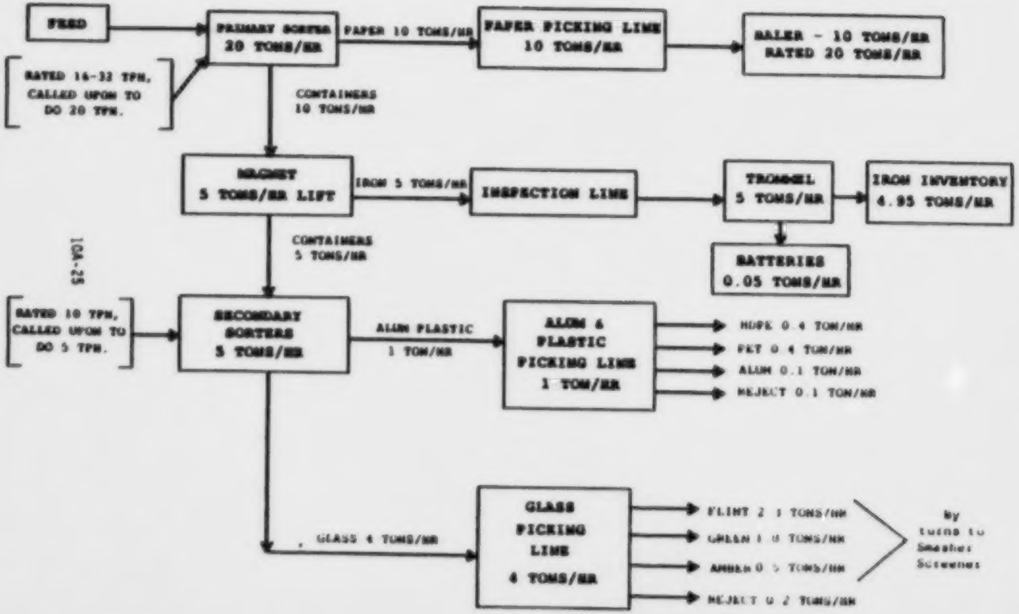
In a well arranged IPF, most of the human workforce will be housed in a sorting room separate from the mechanical equipment. Dust and noise are excluded by a combination of walls and ventilation systems. Another advantage of a separate sorting room is ease of control of lighting, temperature, and sound level.

By using hoods, blowers and a central baghouse, dusty environments can be confined to a portion of the tipping floor and the immediate surroundings of the size-reduction devices.

Conclusion

The intermediate processing treatment of household recyclables promises to help recycling become a major contributor to the solution of the solid waste management problem. In terms of the three lessons learned from earlier types of recycling efforts, it will facilitate the quality improvement, and therefore the probable sale, of post-consumer materials; it will industrialize the process of sorting recyclables; and it will enable the accumulation and collection of recyclable materials to take place in a simplified and convenient manner.

OGDEN MARTIN SYSTEMS, INC.
MATERIALS RECOVERY FACILITY
Flow Diagram


FEED
RATED 16-32 TPH, CALLED UPON TO DO 20 TPH.
PRIMARY SORTER 20 TONS/HR
PAPER 10 TONS/HR
PAPER PICKING LINE 10 TONS/HR
BALER - 10 TONS/HR RATED 20 TONS/HR
CONTAINERS 10 TONS/HR
MAGNET 5 TONS/HR LIFT
IRON 5 TONS/HR
INSPECTION LINE
TROMMEL 5 TONS/HR
IRON INVENTORY 4.95 TONS/HR
BATTERIES 0.05 TONS/HR
CONTAINERS 5 TONS/HR
RATED 10 TPH, CALLED UPON TO DO 5 TPH.
SECONDARY SORTERS 5 TONS/HR
ALUM PLASTIC 1 TON/HR
ALUM & PLASTIC PICKING LINE 1 TON/HR
HDPE 0.4 TON/HR
PET 0.4 TON/HR
ALUM 0.1 TON/HR
REJECT 0.1 TON/HR
GLASS 4 TONS/HR
GLASS PICKING LINE 4 TONS/HR
FLINT 2.3 TONS/HR
GREEN 1.0 TONS/HR
AMBER 0.5 TONS/HR
REJECT 0.2 TONS/HR
by
Turns to
Smasher
Screener

Should we junk Bulky trash or recycle it

by Philippe VACHETTE
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73230 SAINT-ALBAN-LEYSSE
FRANCE

A - No more garbage dumps ! What can we do with the trash mountains ?

Chambéry aera, June 1985 : the mixed-trash garbage dump is full and must be closed ; this is a situation common to many other cities in Europe...
For many months now, the authorities have been fighting against local opposition, over the differing versions of a new project : "the Burying Center". But for the time being, what can be done about the ordinary person's bulky trash ; those mountains of household waste not collected by the traditional garbage services ?

A solution to the problem -"Voluntary Depot Centers" or "Recycling Centers"- (a solution highly promoted by the National Agency for Recovery and Disposal of Waste - Agence Nationale pour la Récupération et l'élimination des Déchets), has been proposed and adopted by the Association of Intermunicipal Authorities in Chambéry (SIAC), representing 14 districts (110 000 inhabitants).

A Recycling Center site, 1 500 - 2 000 m^2, correctly equipped for an urban area (rinforced concrete platforms, tarmac surfaces, wire enclosures) and including containers for non-recycled matter, costs from $ 120 000 - $ 170 000, excluding purchase of land. Realistic operating costs for a center open 7 days a week are approximately $35 000 - $ 40 000 per year, persite plus $10 000 - $ 13 000 per year to cover evacuation of non-recyclable matter.
The Recycling Centers have now been in operation for 3 years and we are now able to look at some results and comment about future possibilities.

B - The service offered by the Recycling Centers in Chambéry :

Since 1986, the city has equiped 3 centers with attendants, open every day of the week for the reception of bulky or special household waste. Design and plan were made with TRIVALOR inc Raised platforms give the public, tradesmen and craftsmen easy access to containers ; they can, if necessary, get help from the operator-attendant of the Center. Wastes are sorted in 20 categories such as rubble, branches and cut grass, burnable waste, lightweight iron, heavy iron, cast iron, glass, paper, cardboard, tires, batteries, used automobile oil, non-ferrous metals, bric-à-brac, wood, plastic.

C - Mains results

In 1988, from the urban area of Chambery (population 110 000) 33 500 m^3 of waste were brought by the people and sorted into 20 categories. These 20 categories can be grouped into for main classes :

- incineration	35 % of the total volume	
- land fill	23 % " "	
- recycling bulk form	31 % " "	41 % recycling
- carrefully sorted or salvaged	10 % " "	

- The present amount of total recycled wastes is 14 000 m^3 41 % of the volume, the aim Ist 50 % in 1992.
- Recycling + incineration represents 77 % of the total volumes, the aim being 90 % for 1992 and increase the life of the land fill times for.

In France were environmental concern of the general public is not high one third of the total amount of waste will be illegally dumped if there are no recycling centers available.

If the cost of cleaning upon illegal dump is $ 15 per cubic meter, then the amount spend to remove and clean the illegally dumped trash will reach 165 000 $/year (see the following page).

D - To recycle is always -in the end- to gain

Such is the viewpoint expressed systematically by TRIVALOR Inc., the Recycling Center operators with whom the SIAC has made an agreement. Yet the general and seemingly long term decrease in the price of raw or reprocessed raw materials tends to encourage a "laissez-faire" or "junk it, don't sort it" attitude.
Overall consideration of the local economy invalidates this theory !
Let us take three examples of waste sorted by TRIVALOR instead of being burned or junked :

a) - Paper and cardboard :

If they are not extracted, 95 % of paper and cardboard goes into household waste. In actual fact, the cost of collecting this waste is approximately $50 - 80 per ton* and the cost of incineration $20 - 25 per ton. At the present time, the three Recycling Centers collect 15 - 22 ton of paper per month and 3 - 5 ton of cardboard ; these are collected and recycled in bulk (in 1988 "sale value" = 550 $ per month ie - waste removed for recycling at no cost). This is therefore an indirect savings of $500 - 1 000 per month for the local community.
Even if "sale value" = 0, it is worth sorting paper, cardboard and newspapers entering the Recycling Centers rather than leaving them with household waste

b) - Wood

At the present time, waste wood makes up more than one tenth of the volume incinerated. Because Chambéry is'nt equipped with a heat recuperation system in its incinerators, it is a source of combustion wich is entirely lost.
Therefore the idea of separating wood waste ofsufficientsize (rafters, beams, planks) and then cutting the waste into log for firewood, was suggested. The work is carried out by unemployed adults in financial difficulty supervised by humanitarian organizations. Profits are added to the basic unemployment allowances already distributed to these unemployed people.

So, even when a solution "sale value = 0", is obtained, when waste cannot be sold, it is better to procces it and certainly more ecologically sound to be sorted and recycled than simply dumped.

The Chambery experience shows in all evidence that a recycling program can be developed at a very favorable ecological, technical and social level and in addition jobs or alternative employment.
The present price collapse of reprocessed products totally prohibits a compagny from living solely off the sale of recyclable products.

c) Plastic waste in sheet

TRIVALORinc, in association with a inglenieriing office and a university research laboratory have received a patent to recycle flexible plastics waste in sheet even soiled : this patent is used in the construction of publics works on soft soils were it is necessary to reduce the weight of the road materials. In'89, 500 T of plastic waste will be reused with this process, and 2 200 T in'90. 5 positions will be created with this new process which promises to have a very large market.

E - Communication is the main thing !

When talking about managing waste, it is as important to keep a site clean as it is to maintain an active relationship with potential users ; as soon as the project has been approved and voted, it is essential to state :

- what will be done
- why it will be done and the aims
- what people should do.

Without making the effort to inform and explain, facts become distorted to the point of inciting a primary reaction of rejection. Following a series of well-organized and widely-spread false rumors, the opening of the second Recycling Center, south of the city, believed to be a garbage dump by the general public, was delayed four months as the result of a petition over 900 signatures collected in just three weeks !

This is why a multi-media campaign -although belated- was launched in the urban area of Chambery : posters 4m x 3m, information in bus shelters, small bills, radio announcements, inserts in the press, press articles, pamphlets delivered to every mail box. For this publicity campaign, the SIAC had no hesitation in spending a sum equivalent to one site's yearly operation costs.

Conclusion :

Now the system of Recycling Centers is well known to the public and is not only accepted but also appreciated. Especially when it comes to waste, local authorities are often "bound hand and foot" by old habits or by the prejudicies of citizens or professionnals who have voted them into power.

It is why traditionnal expensive and wasteful solutions such as : dumps, door-to-door collection, fixed containers for bulky waste in urban areas, are so difficult to abandon.

This reluctancy to change is generally swept away after having visited a Recycling Center.
Thus "junking" can become a true act of citizenship using a convenient clean and practical site : the Recycling Center.

February 1989

Philippe VACHETTE
Manager of TRIVALOR Inc.

ECONOMIE GLOBALE GENEREE PAR LES DECHETTERIES

- Exemple à partir d'une agglomération de 110 000 hab. Chambéry/Savoie -

- 1988 -

POSTE	SANS RESEAU DE DECHETTERIES	Coût	AVEC RESEAU DE DECHETTERIES	Coût
DEPOTS SAUVAGES	(1) 1/3 VOLUMES COUT DE RESORPTION 95 F/m³	1 060 000	5% = 1 700 m³	161 500
COUT DE MISE EN DECHARGE CONTROLEE	2/3 DES VOLUMES = 22 100 m³ COUT 30 F/m³	663 300	23% = 7 700 m³ + BROYAGE VEGETAUX	231 000 100 000
COUT DE TRANSPORT	2/3 = 22 100 m³ : 30 m³ = 737 voyages X 700 F (DECHARGE CLASSE II - GRENOBLE OU SATOLAS)	515 900	23% = 7 700 m³ : 30 m³ = 257 x 320 F EN DECHARGE CAT. III PROCHE DE LA VILLE	82 240
INCINERATION	0		36% = 12 000 m³ 12 000 m³ x DENSITE 0,11 x 128 F/T = + TRANSPORT $\frac{12\ 000}{30}$ = 400 voy. x 320 F = L'INCINERATEUR EST PROCHE DE LA VILLE A LA DIFFERENCE DES DECHARGES CONTROLEES	168 960 128 000
RECETTE RECYCLAGE	0		11 000 F/mois POUR LE RECYCLAGE VRAC + VALEUR AJOUTEE PAR EMMAUS OU TRIVALOR (BROCANTE, BOIS, PLASTIQUES, NON-FERREUX...) (+ SALAIRES POUR 10 EMPLOYES +)	132 000 240 000
COUT DE FONCTIONNEMENT DES DECHETTERIES	0		FONCTIONNEMENT ENTRETIEN + DIVERS	604 900 90 000
TOTAL		2 239 200		184 500

ECONOMIE THEORIQUE REALISEE : 1 054 700 F - SOIT. 47 % SUR LA SOLUTION DECHARGE CONTROLEE

SI LA DEPENSE APPARENTE POUR LES COLLECTIVITES LOCALES N'EST PAS DE 2 239 000 F HT, C'EST PARCE QUE LES DECHETS GENERENT DES COUTS NON PRIS EN COMPTE PAR LA COMPTABILITE PUBLIQUE OU SUPPORTES PAR DES ACTEURS DIFFERENTS DE LA SEULE COLLECTIVITE.

(1) EN L'ABSENCE D'UN RESEAU DE LIEU DE DEPOT PROCHE ET PRATIQUE, ON ESTIME GENERALEMENT A 1/3 LES VOLUMES DE DECHETS ENCOMBRANTS JETES EN DEPOTS SAUVAGES.

Two-Stage DeNOx Process Test Data from Switzerland's Largest Incineration Plant

by:		
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ABSTRACT

A new process for NOx removal using two boiler zones for chemical injection has been installed and tested at KVA Basel, the largest municipal solid waste (MSW) incineration plant in Switzerland. Urea and methanol are used in combination with proprietary injection nozzles to provide NOx removal performance between 65% and 80% over flue gas temperatures between 1500 and 1900 F. Methanol is used to provide substantial reduction of ammonia slip and/or air preheater deposits. Between 1740 an d 1830 F at 65% NOx removal, ammonia slip of 2.5 to 5.0 ppm (without methanol) can be reduced to less than 1.0 ppm. Further development includes award of a contract to install the Two-Stage DeNOx Process on a 325 MW coal-fired boiler in West Germany. The startup date is in August, 1989, and we plan to provide additional details in a forthcoming paper.

PLANT AND DeNOx PROCESS EQUIPMENT DESCRIPTION

There are four boilers at the KVA Basel incineration plant. Two are 46 years old (each rated at 220 TPD) and two are 24 years old (each rated at 330 TPD). The Two-Stage DeNOx Process was installed on one of the larger boilers. Figure 1 provides a furnace system cross-section which shows the two zones of chemical injection, the flue gas flow path, and the location of the temperature sensor used for process control. Urea is provided in bulk granular form and mixed with water prior to being pumped to proprietary injection nozzles in the first zone. Methanol is supplied by liquid pumps to the second-zone injector nozzles. The flowrates of chemical(s) to the various injection locations are controlled by a single flue gas temperature sensor. The flue gas flowrate is 52,000 DSCFM at 11% O2 and the uncontrolled NOx level is 160 dppm. In order to provide a high degree of flue gas mixing, injector system airflow totalling 1500 SCFM is used.

WIDTH OF THE TEMPERATURE WINDOW

Due to wide variations in burning characteristics of the solid waste, the flue gas temperature (measured by thermocouple upstream from the first injection zone) varies between about 1400 and 1900 F. The control system compensates for much of this temperature fluctuation by changing the location(s) where the chemical(s) are injected. As shown in Figure 2, using a molar ratio of urea/NOx of 0.80 plus methanol/NOx of 0.14 produces 80% NOx removal in the KVA Basel boiler at flue gas temperatures between 1600 and 1850 F. The data in Figure 2 was collected at 5-second intervals and subsequently averaged for a 24-hr period of operation. Scatter in the averaged values are due to rapid temperature fluctuations in the KVA Basel furnace. The wide temperature window for high levels of DeNOx is due to (a) a high degree of flue gas mixing provided by proprietary injector nozzles, (b) operation of the temperature control system to adjust the location(s) where chemical(s) are injected, and (c) the drying time for urea droplets, which provides some delay and allows urea dispersion before DeNOx reactions begin to occur. We have called this delay time phenomenon the "popcorn effect".

DATA ACQUISITION

Extreme care was taken to obtain repeatable and representative data during a two-month testing program. For example, NO signals were obtained using two separate gas analyzers: one chemiluminescent type and one NDIR type. In the same way, duplicate signals were obtained for O_2 and CO, and all signals, including boiler and process operating parameters, were recorded using a computerized data-logging system. Ammonia slip was also determined by two different methods: (a) continuous NDIR gas analyzer (operating at 500 F), and (b) 15-minute grab samples analyzed by wet chemistry. It was found that both methods of ammonia sampling provided consistent and representative results.

UREA/NOx MOLAR RATIO

Figure 3 shows the effect of urea/NOx molar ratio on percent NOx removal. DeNOx levels of 78%, 65%, and 48% are achieved at urea/NOx molar ratios of 0.80, 0.60, and 0.40, respectively. The corresponding percentages of urea utilization are 49%, 54%, and 60%. By definition, a urea/NOx molar ratio of 0.50 equals 1.0 moles of NH_2 per mole of NOx. These results are believed to be representative for electric utility boilers where the flue gas temperature at any given injection location can be maintained in a desired range from about 1740 to 1830 F.

AMMONIA SLIP REDUCTION USING METHANOL

One of the important features of the Two-Stage DeNOx Process is the use of methanol to reduce ammonia slip and related concerns including air-preheater deposits. At KVA Basel, relatively high levels of ammonia slip (i.e. 15 to 40 ppm at urea/NOx ratio of 0.80) can be tolerated since the ESP system operates at 460 F and there is very little deposition of ammonia salts, either on equipment surfaces or ESP ash. For coal-fired boiler applications, however, such high levels of ammonia slip are unacceptable. Figure 4 shows ammonia slip as a function of flue gas temperature, both with and without methanol injection at a urea/NOx ratio of 0.62, equivalent to 65% NOx removal. As can be seen, when methanol is added, very large reductions in ammonia slip are realized. Between 1740 and 1830 F at 65% NOx removal, ammonia slip of 2.5 to 5.0 ppm (without methanol) can be reduced to less than 1.0 ppm. In coal-fired boiler applications, this would be critically important for minimizing air preheater deposits and/or, when required, ammonia contamination of ESP ash. The Two-Stage DeNOx Process can thus be optimized to meet site-specific requirements.

OPERATIONAL CHARACTERISTICS AND DISCUSSION

Despite rapid flue gas temperature fluctuations of up to 100 F per minute observed at KVA Basel, the Two-Stage DeNOx Process provides relatively uniform, reduced levels of outlet NOx concentration. In Figure 5, typical strip chart records of gas analyzer NOx signals show variations from 150±40 ppm (without chemical injection) to 40±20 ppm (with chemical injection).

The 160 dppm NOx level at the KVA Basel boiler is relatively low. In higher NOx level applications, the Two-Stage DeNOx Process is expected to operate with (a) lower urea ratio, and/or (b) reduced injector system flowrate at a given DeNOx performance level. It should also be noted from Figures 2, 3, & 4 that ammonia slip increases sharply whenever DeNOx is increased above 65%. Whether or not this is an unavoidable consequence of fundamental NHi-radical chemistry remains an open question.

In the KVA Basel boiler, there is a relatively high degree of stratified flow which requires that high levels of mixing be provided by the injection system. This is why the injector airflow requirement is as high as about 2.9% of the total flue gas mass flowrate. Typical electric utility boilers have a much higher level of natural turbulence, so that the injector system flow requirement is less. For example, typical designs for utility boilers show that the Two-Stage DeNOx Process achieves performance levels similar to those observed at KVA Basel with an injector system flow requirement between only 0.6% and 1.0% of the flue gas mass flowrate.

Proprietary injector nozzles are an important part of the Two-Stage DeNOx Process. Boiler flow models and computer simulations are used to optimize the design of specific injection systems. The Two-Stage DeNOx Process is based on simple engineering principles which are applied to specific boiler configurations using sophistocated design techniques. Proprietary chemicals are not used.

Further development includes award of a contract to install the Two-Stage DeNOx Process on a 325 MW coal-fired boiler in West Germany. The startup date is in August, 1989, and we plan to provide additional details in a forthcoming paper.

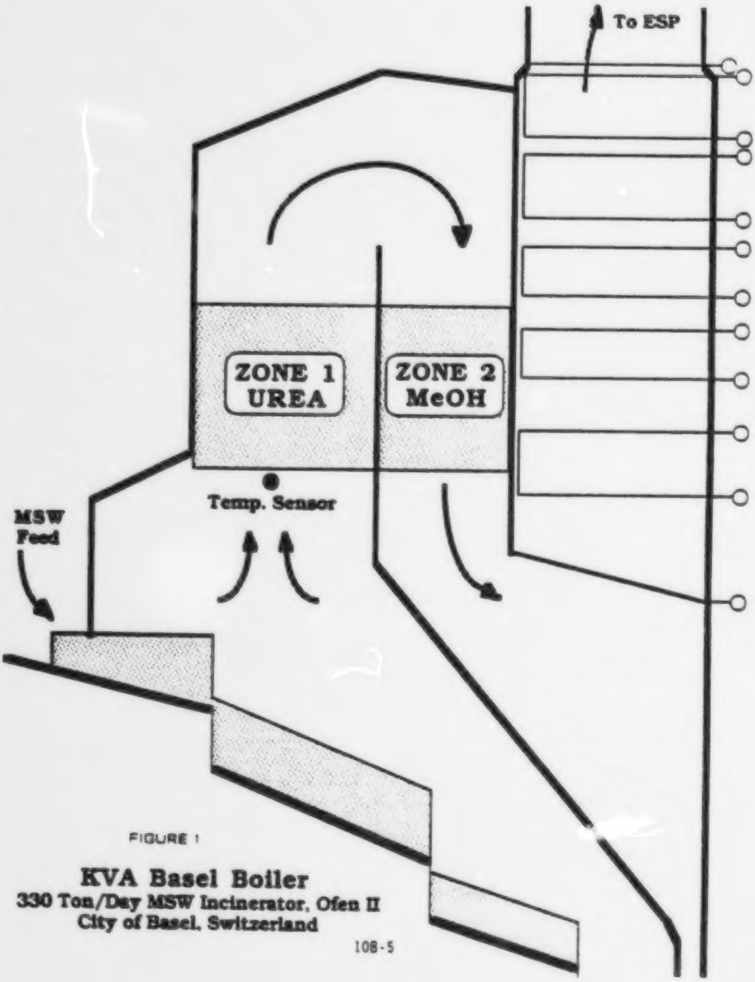
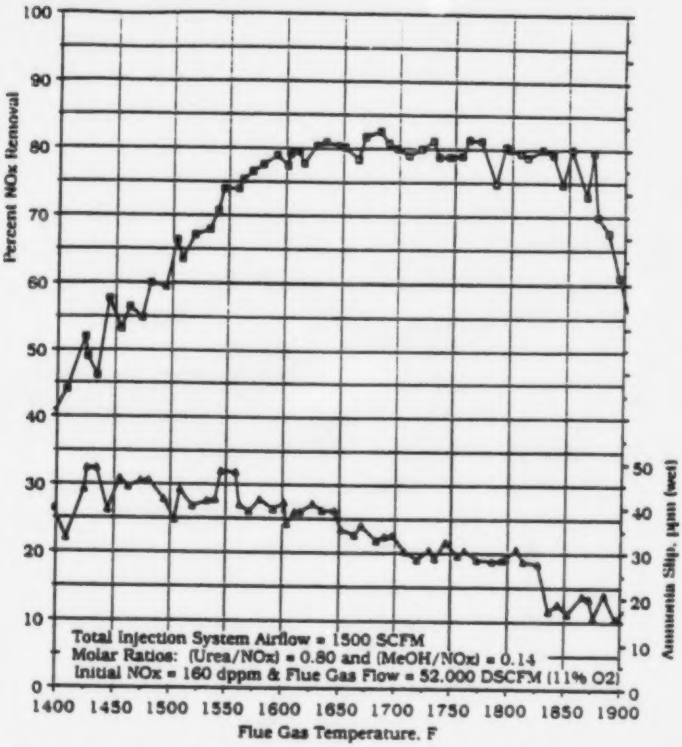

To ESP
ZONE 1
UREA
ZONE 2
MeOH
Temp. Sensor
MSW
Feed

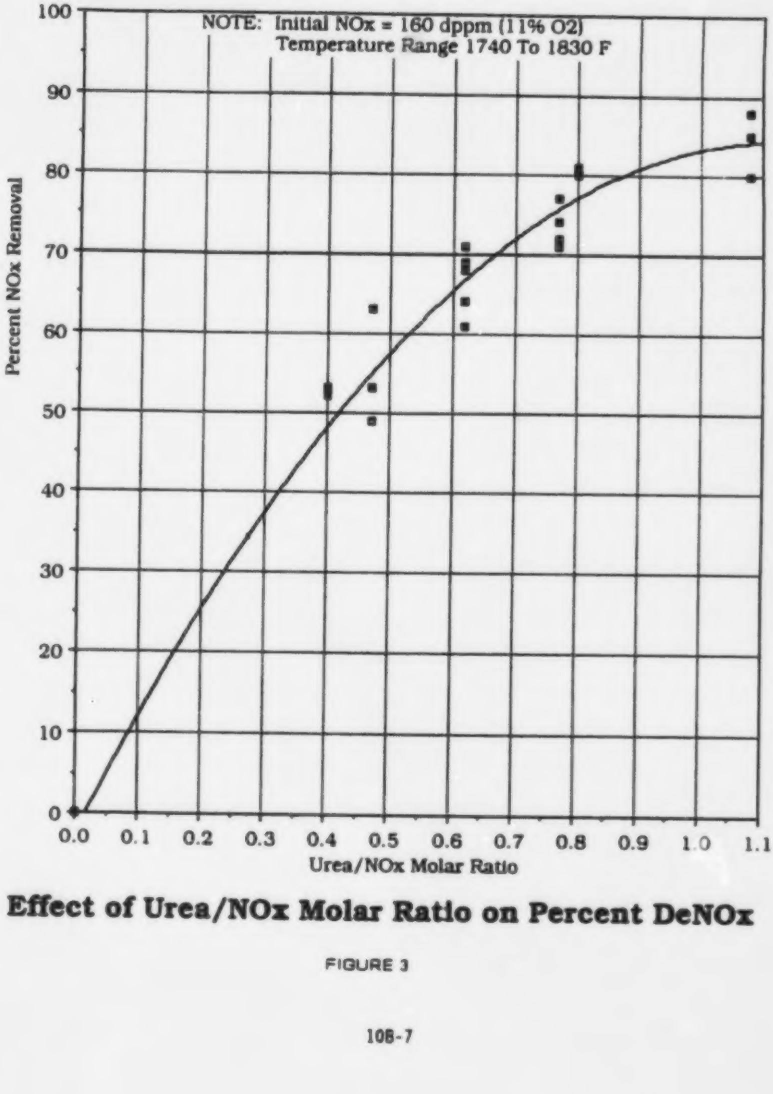
FIGURE 1

KVA Basel Boiler
330 Ton/Day MSW Incinerator, Ofen II
City of Basel, Switzerland



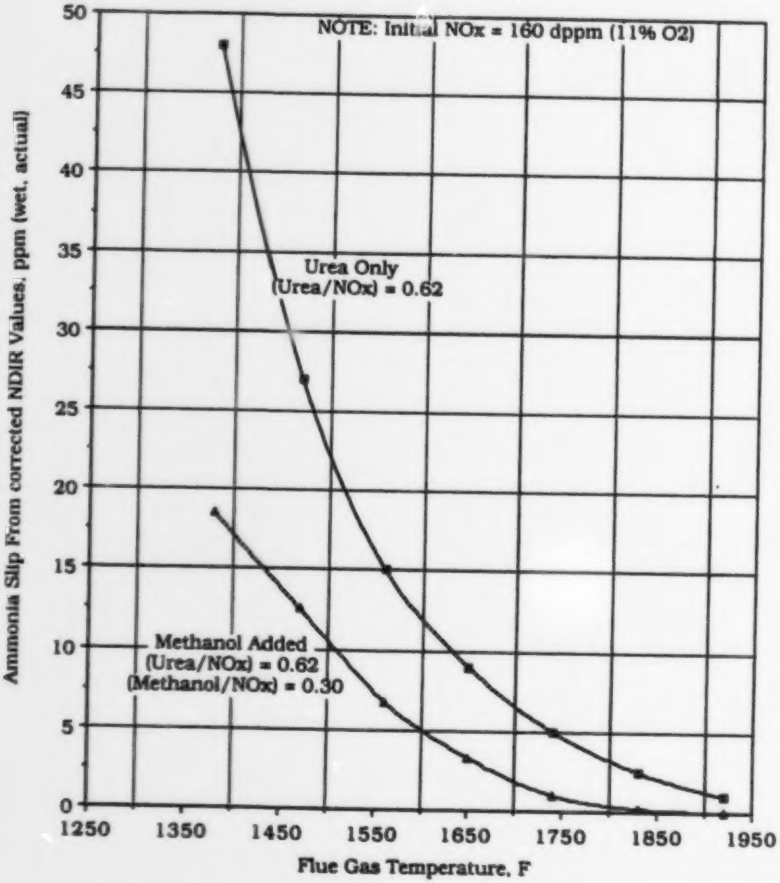
Temperature Window of Two-Stage DeNOx Process

FIGURE 2



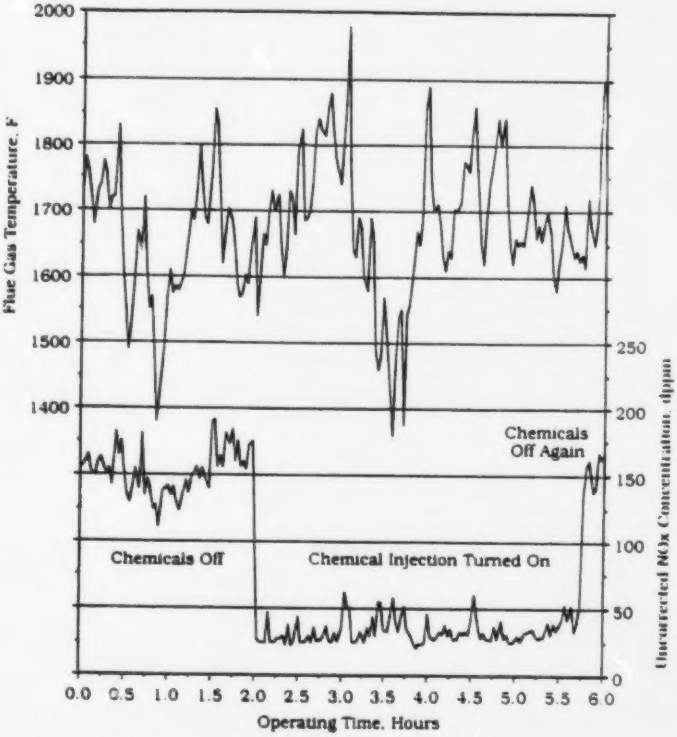
Effect of Urea/NOx Molar Ratio on Percent DeNOx

FIGURE 3



Effect of Methanol Injection on NH3 Slip

FIGURE 4



Operating Characteristics From Chart Records

FIGURE 5

COMBUSTION RETROFIT STRATEGIES FOR CONTROL OF AIR EMISSIONS FROM MUNICIPAL WASTE COMBUSTORS

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ABSTRACT

As part of U.S. EPA's regulatory development effort on municipal waste combustors (MWCs) a program was initiated to develop potential retrofit alternatives for control of air emissions from existing facilities. The alternatives evaluated included combustion modifications and retrofit of flue gas cleaning equipment. A model plant approach was used to address the majority of combustion technologies in the existing MWC population. Baseline emission estimates were made for 12 model plants using the available data for each class of combustors. Retrofit control alternatives were then developed for each model plant. Pollutant emission reductions were estimated for each control alternative, and capital and annual costs are being developed. This paper summarizes the combustion retrofits recommended in the study and documents the basis for emission performance estimates for combustion control alternatives.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for presentation and publication.

BACKGROUND

On July 7, 1987, the U.S. EPA published an advance notice of proposed rulemaking (ANPRM) describing its assessment of air emissions from municipal waste combustors (MWCs) (1). The ANPRM announced EPA's intent to develop air regulations for new and existing MWCs under Sections 111(b) and 111(d) of the Clean Air Act, respectively. The use of Section 111(d) requires that guidelines be issued for existing sources in the form of an emission limit, and that states submit implementation plans to EPA describing action that will be taken to ensure that existing facilities comply with the guidelines. Considerations such as the remaining useful life of a facility may be included in an implementation plan. The responsibility for final approval of each plan rests with EPA. The schedule for the rulemaking includes simultaneous proposal of the new source performance standard (NSPS) and issuance of draft guidelines for existing sources in November 1989. Promulgation of the NSPS and issuance of final guidelines are scheduled for December 1990, with state plans due for existing sources in September 1991.

An extensive data and information gathering effort preceded publication of the ANPRM, the results of which were published in a nine volume Report to Congress (2). However, the subject of existing source retrofit, which may result from the 111(d) rulemaking, was not specifically addressed in the Report to Congress. Retrofit actions involve case by case evaluation of issues such as technical feasibility and costs. Cost effectiveness is an input to regulatory decision making in Section 111 standards. Information on the technical feasibility and economic impacts of MWC retrofits posed a major data gap in the regulatory development effort.

A study was initiated by EPA's Offices of Research and Development (ORD) and Air Quality Planning and Standards (OAQPS) to fill some of the data gaps associated with potential MWC retrofits. The goals of the study were to provide an estimate of the baseline emission performance for the existing population of MWCs, to examine the need and technical feasibility of applying various retrofit controls, and to develop the cost impacts and emission reductions associated with each retrofit control alternative. Because it was impossible to evaluate retrofit options for every existing MWC individually, EPA used model plants to represent segments of the existing MWC population. The model plants were developed using available information related to the design and operating characteristics of each subgroup of combustors in the existing population.

SELECTION OF MODEL PLANTS

The population of existing MWCs consists of a wide range of combustor design types with varying operating practices. For example, some units burn unprocessed waste on grates while others burn refuse-derived fuel (RDF) in spreader stoker boilers. Nearly all new facilities generate steam, while the majority of older units are used strictly for waste volume reduction. The types of flue gas cleaning equipment employed in the existing population also vary widely. Each existing facility has unique characteristics that can only be evaluated on a site-specific basis. With three decades of technology development and more than 150 facilities in the existing population, the potential for widely variable individual situations clearly exists.

Twelve model plants were developed to represent the existing MWC population. Some existing units with unique designs and few operating units (e.g., fluidized bed combustors) are not represented by a model plant. The combustor design categories included in the study were mass burn refractory wall incinerators (no heat recovery), mass burn waterwall facilities, refuse-derived fuel (RDF) fired spreader stoker boilers, modular starved air facilities, modular excess air systems, and mass burn rotary waterwall MWCs. More than one model was included for refractory wall, mass burn waterwall, RDF, and modular starved air MWCs, since unit sizes or design characteristics vary considerably within each category. Table 1 summarizes characteristics of the 12 model plants.

Site visits were made to actual operating facilities approximating each model plant configuration to gather supporting information on design and operating practices. The plants included in the site visits are identified in Table 2. It is important to distinguish the model plants from the plants that were included in the site selection. Although in many cases the models are similar to the actual facilities visited, the application of a retrofit alternative at an actual plant may require site-specific considerations that were not addressed in the model study.

A second source of supporting information on model plant design and operating practices came from facility responses to an EPA questionnaire that was mailed to all operating MWCs. The responses were used to determine if selected model plant parameters were prevalent throughout each subcategory of combustors.

DEVELOPMENT OF BASELINE EMISSIONS

It was necessary to establish baseline emission estimates for each model plant in order to characterize the performance of the existing MWC population. The pollutants of concern evaluated in the study included chlorinated dibenzo-p-dioxins and chlorinated dibenzofurans (CDD/CDF), carbon monoxide (CO), particulate matter (PM), hydrogen chloride (HCl), and sulfur dioxide (SO_2). Table 3 summarizes the baseline emission levels assigned to each model plant.

The CDD/CDF, CO, and PM emissions data were established after careful review of the existing data base, which included results from compliance and parametric tests. Where no available data existed to characterize the performance of a model plant, engineering judgments were made based on facility design and operation. For example, mass burn refractory wall MWCs were assumed to have higher particulate carryover than waterwall facilities because they typically operate at higher excess air levels. Where a wide scatter of data existed for a class of combustors, a reasonable worst case (upper bound) approach was used to assign baseline emission values. Baseline acid gas emissions were established based on an assumed conversion of chlorine

TABLE 1. MWC MODEL PLANTS

MODEL PLANT TYPE	NO. OF UNITS	UNIT SIZE (tpd)	UNIT SIZE (Mg/day)	AIR POLLUTION CONTROL DEVICE
Refractory Wall/Traveling Grate	2	375	341	Electrostatic Precipitator
Refractory Wall/Rocking Grate	2	120	109	Water Sprays
Refractory Wall/Rotary Kiln	3	300	273	Electrostatic Precipitator
Large Mass Burn Waterwall	3	750	681	Electrostatic Precipitator
Mid-Size Mass Burn Waterwall	3	360	327	Electrostatic Precipitator
Small Mass Burn Waterwall	2	100	91	Electrostatic Precipitator
Large RDF	2	1000	909	Electrostatic Precipitator
Small RDF	2	300	273	Electrostatic Precipitator
Modular Starved Air	3	50	45	Electrostatic Precipitator
Modular Starved Air	2	25	23	None
Modular Excess Air	2	100	91	Electrostatic Precipitator
Rotary Waterwall	2	250	227	Electrostatic Precipitator

TABLE 2. FACILITIES INCLUDED IN SITE VISITS

FACILITY	TYPE	NO. OF UNITS	UNIT SIZE (tpd)	UNIT SIZE (Mg/day)	START-UP DATE
Philadelphia, PA	Refractory Wall/Traveling Grate	2	375	341	1957
Sheboygan, WI	Refractory Wall/Rocking Grate	2	120	109	1965
Dayton, OH	Refractory Wall/Rotary Kiln	2 1	300 300	273 273	1970 1988
Saugus, MA	Mass Burn Waterwall	2	750	681	1975
Nashville, TN	Mass Burn Waterwall	3	2 @ 360 1 @ 400	327 364	1974 1986
New Hanover County, NC	Mass Burn Waterwall	2	100	91	1984
Niagara Falls, NY	RDF Spreader Stoker	2	1000	909	1981
Albany, NY	RDF Spreader Stoker	2	300	273	1981
Tuscaloosa, AL	Modular Starved Air	4	75	68	1981
Waxahachie, TX	Modular Starved Air	2	25	23	1982
Pittsfield, MA	Modular Excess Air	3	120	109	1981
Bay County, FL	Rotary Waterwall	2	250	227	1987

TABLE 3. BASELINE EMISSIONS - COMBUSTOR OUTLET

COMBUSTOR TYPE	CDD/CDF (ng/dscm)	CO (ppmv)	PM (mg/dscm)	HCl (ppmv)	SO_2 (ppmv)
Mass Burn Refractory Wall Model Plants (3)	3500	500	6900 (3 gr/dscf)	500	200
Mass Burn Waterwall - Large	500	50	4600 (2 gr/dscf)	500	200
Mass Burn Waterwall - Midsize	200	50	4600	500	200
Mass Burn Waterwall - Small	2000	400	4600	500	200
RDF Spreader Stoker Model Plants (2)	2000	200	9200 (4 gr/dscf)	500	300
Mass Burn Modular Starved Air Model Plants (2)	400	100	345 (0.15 gr/dscf)	500	200
Mass Burn Modular Excess Air	200	50	4600	500	200
Mass Burn Rotary Waterwall	2000	100	4600	500	200

and sulfur in the waste feed. These emissions are highly variable due to changing feed composition, and the baseline numbers represent nominal values. As new measured data emerge from recently completed emission tests, some revision to the baseline CDD/CDF and CO emissions may be warranted. However, based on review and analysis of the available data on hand in mid-1988, the numbers in Table 3 were established for use in this study.

The pollutant concentrations presented in Table 3 are assumed to be measured at the outlet of the combustor, upstream of flue gas cleaning equipment. The concentrations were adjusted based on assumed flue gas cleaning equipment performance to obtain stack baseline emissions. Two important assumptions were made in order to establish baseline emissions of CDD/CDF at the stack. First, it was assumed that a hot ESP [operating temperature above 450°F (282°C)] would increase CDD/CDF concentrations by 50 percent over the inlet value, resulting in a higher stack concentration. This assumption was made based on results from two full scale data sets (3, 4). CDD/CDF formation reaction rates are influenced by a number of factors, including temperature, residence time, PM concentration, and catalysts. The CDD/CDF formation phenomenon was the subject of a full scale parametric emissions testing program performed by U.S. EPA in late 1988. The results of this testing program may influence this assumption when they become available.

The second assumption affecting baseline emission estimates was that a cold ESP [operating temperature below 450°F (282°C)] did not provide any removal of CDD/CDF. This assumption is a conservative one, because full scale data indicate that CDD/CDF removals approaching 70 percent have been achieved by some ESPs (5). CDD/CDF are routinely found in ESP fly ash samples at varying concentrations, indicating that there is some level of CDD/CDF removal associated with particulate removal. However, the scatter in available ESP efficiency data precluded selection of a CDD/CDF removal efficiency for cold ESPs, and the assumption of zero removal for a worst case analysis.

GOOD COMBUSTION PRACTICE

The design and operation of each model plant in the retrofit study was evaluated against a set of criteria that collectively defines "good combustion practice" (GCP). For this study, the goal of GCP was directed primarily toward minimization of CDD/CDF and other trace organic emissions. The effects of combustion modifications on levels of other pollutants of concern such as NO_x and metals were also considered in the evaluation, although results were not quantitatively presented in all cases. The goals of GCP as directed toward the control of organic emissions are to:

- maximize thermal destruction of organic pollutants in the furnace through efficient mixing of air and fuel at appropriate temperatures
- minimize the potential for formation of CDD/CDF in lower temperature portions of the MWC system

The GCP control strategy includes a number of components that contribute to minimizing the emission of trace organic compounds. These components are presented in Table 4. Each component is considered to be necessary to ensure that the two main GCP goals are met, yet no single component is sufficient by itself to ensure that this occurs. The specific values for each component vary according to technology and are based either on fundamental data or on system-specific performance data. A more extensive description of the development of the guidelines for good combustion practice is provided in a separate paper presented at this conference (6).

COMBUSTION RETROFIT ALTERNATIVES

The combustion retrofit options recommended for each model plant were applied in an attempt to achieve the combustion guidelines outlined for a specific design. In many cases they were based on successful results from retrofits that have been documented at full scale MWCs in the existing source population. Specific model plant case studies are summarized below.

MASS BURN WATERWALL MODELS

It was judged that the model plants representing the large and mid-size mass burn waterwall facilities met the majority of design and operating criteria that define GCP. The available CDD/CDF emissions data from operating units represented by this model reflect this assessment. The majority of these plants [unit size greater than 250 tpd (227 Mg/day)] use European technology (Von Roll and Martin stokers). Several others employ American technology (Detroit Stoker) and the data available for these units also verify that GCP is in place at most of these facilities. The only recommended modification for these two model plants was installation of continuous CO monitors and readouts for continuous verification of GCP.

The third mass burn waterwall model plant consisted of two combustors, each at 100 tpd (91 Mg/day) capacity, controlled by individual ESPs. The available information on actual operating plants of this size indicated that, while some facilities meet the GCP requirements, others are lacking key design and operating criteria that will ensure low trace organics emissions. The baseline CDD/CDF emissions (2000 ng/dscm, corrected to 7 percent O_2) were established using data measured from the Quebec City, Quebec, MWC (7). The only available CDD/CDF emissions data from a U.S. plant in this subcategory were from the Hampton, VA, facility. Several early tests at Hampton indicated CDD/CDF emissions on the order of 10,000-20,000 ng/dscm (8, 9). These data were not considered to be representative of the performance of this category of combustors and were not used in the baseline emission estimates.

The system components addressed in the model plant case study were the design of the overfire air system, control and distribution of underfire air supplies, and installation of auxiliary fuel burners for use during start-up and shutdown. While auxiliary fuel may not impact emissions on a day-to-day basis, it is considered a necessary component of GCP. It was assumed that the exact configuration of the redesigned overfire air system would be

TABLE 4. COMPONENTS OF GOOD COMBUSTION PRACTICE

Temperature at fully mixed condition

Combustion air (excess air or O_2 in flue gas)

Primary air

Secondary air

Air preheat

Mixing

PM carryover

Operating load

Waste feed conditions

CO in flue gas

Waste burnout

Air pollution control device inlet temperature established as a result of flow modeling studies, and that CO monitors would be installed as a gross indicator of macroscale mixing. An oxygen trim loop was incorporated into the existing combustion control scheme of each unit to automatically adjust underfire air distribution, thus preventing channeling and providing for good waste burnout. It was estimated that the proper application of these modifications would result in an order of magnitude reduction of CDD/CDF emissions to 200 ng/dscm and a reduction of CO emissions to less than 50 ppmv. Results from two successful combustion retrofit programs carried out at the Hampton, VA, and the Quebec City mass burn waterwall facilities form the basis for these estimates.

Quebec City, Quebec

Details of the Quebec City retrofit have been presented at several conferences similar to this one. Major system components addressed in the Quebec City retrofit program included the overfire air system design, the underfire air supply and distribution, and the physical configuration of the combustion chamber (bull noses, etc.) and the combustion control system. The modifications resulted in a reduction in stack CDD/CDF emission levels of more than an order of magnitude. Following the retrofit, CDD/CDF emissions less than 100 ng/dscm and CO emissions less than 50 ppmv were measured during design operating conditions (10).

Hampton, Virginia

The Hampton plant was tested for CDD/CDF on several occasions between 1981 and 1984, and emissions were generally characterized as being much higher than those measured at other mass burn waterwall facilities. In 1984, the total CDD/CDF was reported to be in excess of 20,000 ng/dscm (9). In addition, typical ESP operating temperatures were normally in the range of 550-600°F (287-315°C), where rates of CDD/CDF formation appear to be maximized (11). The successful retrofit of the two 100 tpd (91 Mg/day) units required a number of modifications that addressed both environmental and operational concerns. Although the modifications extrapolated from the Hampton experience and used in the EPA retrofit study were limited to those that directly addressed the GCP guidelines, additional environmental benefits may have been realized as a result of the operational modifications. The retrofits are summarized below.

The initial system start-up was unsuccessful due to problems associated mainly with feeding. These problems were addressed by installing ram feeders which automatically regulate the charge rate in response to steam flow set-points. Excessive corrosion of waterwall tube surfaces, refractory studs, and outside boiler casing steel also occurred in the early stages of operation. The tube stud pattern was tightened up and good seals were established between the boiler tubes and the outside casing to eliminate this problem. In addition, the silicon carbide refractory in the lower furnace was replaced by alloy blocks which improved heat transfer and reduced clinker formation in the lower furnace.

Each unit has a feed section and two grate sections (burning grates and finishing grates). The original grates were cast iron, and burnout was occurring every 4-6 months. The cast iron grates were gradually replaced with chrome-nickel alloy grates and their life was extended to 2-3 years. Continual problems with the water-cooled feed chutes resulted in their replacement with dry chutes.

During periods of heavy rainfall when incoming waste had a high moisture content, the units had serious problems achieving good burnout, and sometimes would smolder. This problem was corrected by installing steam coil air preheaters, which have allowed the units to be operated at full load even when receiving very wet refuse.

The grate speed was originally controlled automatically in response to furnace temperatures, and the operating limits ranged from 40 to 80 percent of full speed. The grate speed control was switched to manual, allowing the operating range to be varied from 0 to 80 percent and providing the operator with better ability to use grate speed adjustment to improve waste burnout conditions.

While the combustor modifications discussed above may have had some effect on air emissions, the more direct reason for their implementation was to address operational problems. The key modifications that led to reduced air emissions were associated with combustion air supply, distribution, and control.

Following the 1984 emissions tests, continuous CO monitors were installed and very high CO spikes were observed. Based on these data, it was clear that deficiencies in air distribution and mixing existed. A review of the overfire air forced draft (FD) fan performance indicated that the fan was supplying less than 50 percent of its rated flow. Tips were added to the fan blades and the size of the discharge duct was increased. In addition, the existing overfire air nozzles were replaced, increasing individual nozzle diameters from 1.25 to 2 in (3.175 to 5.08 cm). These modifications addressed the need for increasing the amount of airflow; the second concern was related to improving mixing.

The units have two rows of overfire air nozzles on each of the front and rear walls. The performance of the overfire air system was evaluated based on visual observations. It was noted that air streams from the lower front and lower rear nozzle rows converged at the top of the burning grate. This resulted in quenching and some clinker formation on the grates. The orientation of the front wall nozzles was raised from -45° to -22.5°, and the rear wall nozzle orientation was raised from -20° to 0° (horizontal). This modification raised the jet convergence point approximately 4.5-5 feet (1.5 m) above the grate. No changes were made to the orientation of the two upper nozzle rows. Additional rows of overfire air nozzles were added on the rear and side walls. These rows are supplied by ducting from the underfire air supplies; however, they are used only periodically at 0-10 percent of full damper opening. The current overfire air operation normally includes only the two front wall rows and the two rear wall rows, which combine to supply 45 percent of the total combustion air.

Three additional action items were identified as a result of a workshop held by project participants. First, it was proposed that a taller stack be installed to increase dispersion of flue gases. The two existing 90 ft (27 m) stacks were removed and a new 248.5 ft (75.7 m) stack with a double flue was installed.

Secondly, it was necessary to increase overall system efficiency. This was accomplished by retrofitting both units with new economizer tube banks which lowered combustor exit gas temperatures from 550°F to 420°F (288 to 216°C). Although the resulting increase in efficiency reduced fuel consumption on an hourly basis, overall availability increased as the result of the system modifications, so annual steam production and waste throughput actually increased. This modification also reduced ESP operating temperatures, thereby minimizing conditions which are thought to promote formation of CDD/CDF in the control device. Initially there was a concern that the temperature reduction would adversely affect particulate collection efficiency. However, emission tests revealed that stack PM emissions were not significantly changed.

Lastly, modifications were required to improve the combustion control system. The main features of the existing control scheme included automatic adjustment of underfire air flows and ram speeds based on steam flow setpoints. The overfire air flowrate is maintained constant. A 15-point CO profile was performed at the economizer outlet, and some preliminary CDD/CDF emissions tests were performed. The results indicated a 98 percent reduction in CDD/CDF emissions, but periodic CO spikes were still occurring. The spikes were judged to be due to sneakage that occurred up the rear furnace wall when the majority of active burning was concentrated on the burning grate. When the bed length was extended so that active burning was occurring on the finishing grate, there were problems with solids burnout. An oxygen trim loop was installed to provide automatic control of the underfire air distribution based on the exhaust gas O_2 concentration. With this control loop in place, the distribution of underfire air is balanced between the burning and finishing grates, providing good waste burnout and maintaining 7-9 percent O_2 in the exhaust gases.

The combustion modifications made at the Hampton MWC resulted in dramatic reductions in stack emissions of CDD/CDF and CO. The most recent emission test results reported for the facility indicated that both of the units achieved CDD/CDF emissions below 200 ng/dscm and CO emissions less than 30 ppmv (corrected to 7 percent O_2) (12). These emission levels are in the range demonstrated by many of the newest state-of-the-art mass burn waterwall facilities and represent an emission reduction of at least two orders of magnitude from previously measured levels.

The results of the Quebec and Hampton retrofit programs confirm that substantial reductions in CDD/CDF and CO can result from the application of combustion modifications. These results provide the rationale for performance assumptions made for the small mass burn waterwall model plant following application of the combustion retrofits. Many of the combustion modifications employed at the Quebec City and Hampton facilities were also adapted for application to other model plants.

MASS BURN REFRACTORY WALL INCINERATORS

Many older refractory wall incinerators that were constructed without heat recovery do not satisfy the recommended GCP guidelines for MWCs. This is not surprising because these systems were designed primarily for waste volume reduction, with little concern for organic emission levels. The three model plants developed in this study included two rectangular refractory designs, one with traveling grates and one with rocking grates, and one system with a split flow design using reciprocating grate sections and a refractory kiln.

Traveling grates are not an acceptable design for mass burn furnaces; while they are well suited for use in RDF spreader stokers, they do not provide bed agitation to promote good burnout of unprocessed waste. One of the retrofits suggested for the traveling grate model plant was grate replacement. The traveling grates were replaced with multiple reciprocating grate sections that included individually controllable underfire air supplies and ram feeders. In 1987, six operating plants were identified in the U.S. using traveling grates. Three of the six have been permanently closed since the retrofit study began.

A recommended modification for the two rectangular refractory wall models was reconfiguration of furnace geometries to include bull noses for increased flue gas residence times and improved mixing. The Philadelphia Northwest incinerator, which is now permanently closed, demonstrated that installing bull noses in the upper furnace and relocating water quench sprays farther downstream of the furnace exit increased gas residence time for mixing, resulting in a six-fold reduction in CDD/CDF in the stack (13).

Other recommended modifications related to combustion airflows included a reduction in excess air operating level and installation of high velocity overfire air nozzle rows. Reducing the excess air level results in higher furnace temperatures and lower particulate carryover. High pressure overfire air nozzles contribute to improved mixing. Recommendations were made that all units install a source of auxiliary fuel for startup and shutdown conditions and that continuous CO and O_2 monitors be in place to verify operation within the bounds of the recommended operating envelope. It was estimated that CDD/CDF emissions could be reduced to 500 ng/dscm for the three refractory wall model plants as a result of these modifications.

MODULAR STARVED AIR MWCs

The population of modular starved air MWCs is characterized by units that can achieve relatively low CDD/CDF, CO, and unabated particulate emissions when operated according to their design. Many existing modular starved air MWCs are not equipped with flue gas cleaning equipment. However, in recent years more modular units have been constructed with PM controls.

During normal operating conditions modular starved air facilities have achieved CDD/CDF emission rates on the order of 300-500 ng/dscm (14). One exception exists for a facility equipped with an ESP operating at a temperature near 600°F (316°C), which reported CDD/CDF emissions in excess of 3000 ng/dscm. It is believed that the higher CDD/CDF emission levels are partially due to formation reactions occurring in the ESP, where particulate concentrations and flue gas residence times are increased.

These results indicate that retrofit of an MWC to include a hot ESP may potentially increase concentrations of CDD/CDF in stack gases. One of the model plants in this category included a hot ESP in its design. The proposed solution for the facility was the addition of a separate economizer module between the waste heat boiler and the ESP. The economizer was designed to lower flue gas temperatures in the ESP to levels where CDD/CDF formation rates are minimized [less than 450°F (232°C)]. In addition to addressing potential air emission concerns, a properly designed economizer can boost overall system efficiency and facility revenues by increasing steam production rates per ton of waste burned. This has been accomplished at the Tuscaloosa, AL, facility, where the revenues from increased steam production are expected to pay for the economizer retrofit in 5 years or less (15).

RDF SPREADER STOKERS

Two RDF fired model plants were included in this study, representing large [1000 tpd (909 Mg/day)] and small [300 tpd (273 Mg/day)] unit sizes. Other than size, the main difference between the two model plants was the design and location of the air heater. It was assumed that the large model plant was equipped with a regenerative air heater downstream of an ESP. This arrangement was found to be typical for many RDF boilers, and it results in a hot ESP with greater potential for CDD/CDF formation. The small RDF model plant was equipped with a tubular air heater and a cold-side ESP.

The available emissions data base indicates that RDF fired systems exhibit higher emissions of CDD/CDF than many other technologies (14). As further research on CDD/CDF formation and destruction is performed, it has been postulated that the increased CDD/CDF emissions may be a result of the higher PM carryover which is inherent in semi-suspension fired systems. Increased carryover of particulate bound organics may lead to higher CDD/CDF formation rates in low temperature portions of the MWC system. Figure 1 illustrates the correlation between PM carryover and CDD for two MWCs (one mass burn waterwall unit and one spreader stoker boiler). While PM carryover cannot be eliminated, other controlling design and operating features (e.g., air distribution) can be addressed to provide conditions that will minimize the contribution of CDD/CDF emissions from PM carryover.

Recommended modifications for both RDF model plants included installation of metered feeding systems to ensure consistent feed rates and uniform waste bed distribution, and optimization of overfire air designs and overfire/underfire air ratios for good mixing. Site-specific conditions may make it impossible to include metered feeding modules at actual operating facilities. Lastly, it was recommended that facilities consider modifying ducting

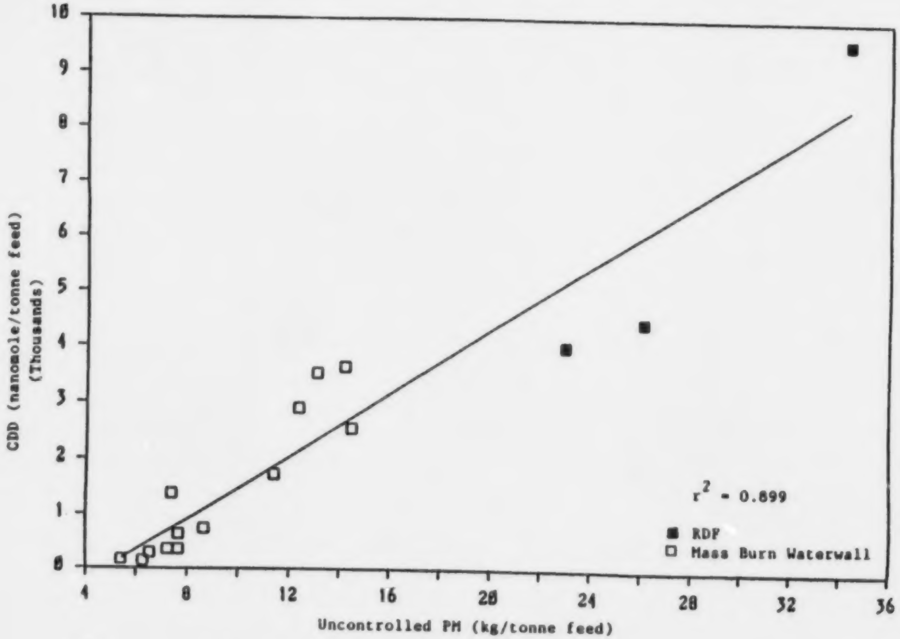

Figure 1. CDD Emissions versus uncontrolled PM.

arrangement to convert hot-side ESPs to cold-side. This application may also be difficult due to site-specific spatial constraints, and additional measures may be necessary for particulate removal from flue gases prior to entering regenerative air heaters.

Although little measured data exist for RDF fired boilers to support the effects of these modifications on air emissions, it was estimated that baseline CDD/CDF emissions could be reduced from 2000 to 500 ng/dscm in the stack. One existing RDF facility has demonstrated that derating can have a beneficial effect on CDD/CDF emissions (16). However, this is seen as a last choice option if more economical modifications are unable to reduce emissions to desired levels. It is still necessary to optimize mixing at temperature and minimize low temperature CDD/CDF formation pathways even when derating is considered.

SUMMARY

EPA's model plant MWC retrofit study concludes that significant reductions in trace organic emission levels can be achieved through the proper application of combustion retrofits. The utility of combustion modification as a control measure for reducing CDD/CDF emissions has been demonstrated in several MWC retrofit programs. Although any retrofit application will encounter site-specific limitations that cannot be accounted for in a model plant study, it is clear that combustion retrofit options merit serious consideration as a control strategy for existing MWCs subject to more stringent CDD/CDF emission standards. The retrofit control alternative selected by an operating facility (if one is necessary) will ultimately be decided by two factors. First, there must be assurance that required emission limits will be achieved by the chosen control alternative. Secondly, if two alternatives provide the facility with the ability to meet the existing source standard, costs will dictate the selection of the preferred control alternative.

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MEDICAL WASTE INCINERATION PRACTICES IN MUNICIPAL WASTE COMBUSTORS

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ABSTRACT

A task is currently underway to examine the acceptability of incinerating medical waste in municipal waste combustors (MWCs) from an emission standpoint and to define the necessary conditions and practices to minimize the threat to human health and the environment. This paper discusses the information gathered to date, defines and characterizes types of medical waste, discusses the impacts of burning medical waste on combustor emissions, and outlines important handling and operating considerations. Facility-specific design, handling, and operating practices are also discussed for those MWCs that reportedly accept medical waste. There are very limited data available on the emission impacts associated with combustion of medical waste in MWCs. More research is needed to fully evaluate the impacts on acid gas, dioxin, and metals emissions as well as the design and operating requirements for complete destruction of hazardous solvents, cytotoxic chemicals, and pathogens.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for presentation and publication.

INTRODUCTION

The U.S. EPA is currently developing air emission standards for new and existing municipal waste combustors (MWCs) under Section 111 of the Clean Air Act. In addition, as part of the Medical Waste Tracking Act of 1988, EPA must prepare a comprehensive report to Congress identifying sources, handling procedures, transportation, treatment, and disposal of medical waste. This report must also include an assessment of the present and potential threat that medical waste represents to human health and the environment. It was determined in the MWC information gathering effort that a number of MWCs burn medical waste. The impacts of this practice on air emissions are currently not known. Based on these uncertainties, a study is being conducted to examine and define the practice of incinerating medical waste in MWCs and identify potential environmental and worker health problems associated with this practice. This paper discusses the information gathered to date, defines and characterizes types of medical waste, discusses possible impacts of burning medical waste on combustor emissions, and outlines important handling and operating considerations. Facility-specific design, handling, and operating practices are also discussed for those MWCs that reportedly accept medical waste.

MEDICAL WASTE

Medical waste is defined as any type of waste generated by a biomedical institution, including hospitals, medical laboratories, animal experimentation units, and clinics. Two features of medical waste are of key importance. First, the physical and chemical characteristics of the waste are highly variable. The heat content of the waste can vary from a low of 1000 Btu/lb (2.326×10^6 J/kg) for human tissues, organs, and body parts to a high of over 15,000 Btu/lb (3.489×10^7 J/kg) for plastic containers and disposable equipment (1, 2). In addition to heat content, the ash, moisture, chlorine, and metals contents can also vary significantly from batch to batch.

Second, many of the components of medical waste require special attention. These components are:

- Cytotoxic chemicals
- Hazardous chemicals
- Pathogens
- Toxic metals
- Radioactive materials

Medical waste which does not fall into any of the above categories is considered general refuse and is similar to household waste. This waste generally makes up the bulk of medical waste.

CYTOTOXIC CHEMICALS

Cytotoxic chemicals are substances capable of impairing, injuring, or killing cells. These hazardous pharmaceuticals are used in chemotherapy.

Available information indicates that these agents may not be effectively destroyed at temperatures below 1800°F (982°C) (3).

HAZARDOUS CHEMICALS

A number of laboratory solvents found in medical waste are listed as hazardous under the Resource Conservation and Recovery Act (RCRA). Included among these hazardous solvents are:

Acetone
2-Butanol
Butyl alcohol
Cyclohexane
Diethyl ether
Ethyl acetate
Ethyl alcohol
Heptane
Hexane
Methyl alcohol
Methyl cellosolve
Pentane
Petroleum ether
2-propanol
Sec-butyl alcohol
Tert-butyl alcohol
Tetrahydrofuran
Xylene

These compounds present concerns due to their own hazardous characteristics as well as their potential to serve as dioxin precursors.

PATHOGENS

Infectious waste is defined as medical waste which is capable of producing infectious disease due to the presence of pathogens of sufficient virulence and exposure routes to the host. (Examples: isolation waste, cultures and infectious agents, human blood, and body parts.) Pathogens are defined as agents capable of causing disease such as a bacterium or virus. Infectious waste represents about 10 percent of the total medical waste stream and generally can be completely destroyed in a well designed and operated incinerator. The destruction of infectious waste is one of the primary reasons incineration is recommended for medical waste.

TOXIC METALS

Medical waste contains toxic metals such as lead, cadmium, and mercury. These metals may be emitted into the air or may be leached from the solid residuals into groundwater. The principal mechanisms for the emission of metals are entrainment of metal-bearing particles or vaporization of the metal and transport in the gas phase from the combustion device. The vaporization escape mechanism is particularly important because the vapor condenses into a fine fume which may be difficult to capture with some air pollution control devices.

RADIOACTIVE MATERIALS

Low level radioactive waste can be present in medical waste. The Nuclear Regulatory Commission (NRC) considers incineration to be an excellent means of disposing of radioactive medical waste (3). Medical waste materials deregulated by the NRC include scintillation vials and research animal carcasses with less than 0.05 microcuries of tritium or carbon-14 per gram. Hazardous waste sites have been reluctant to accept this deregulated material, and incineration has been seen as an important alternative to shallow land burial.

MWCs burning even small percentages of medical waste must be able to accommodate highly variable waste compositions and heat content as well as many of the components discussed above.

IMPACT OF MEDICAL WASTE ON COMBUSTOR EMISSIONS

Limited information is available on the emission impacts associated with combustion of medical waste in MWC facilities. However, these impacts can be qualitatively discussed based on the composition of medical waste and emissions data from medical waste incinerators.

Acid gas emissions are a primary concern. Halogenated polymers and copolymers make up a significant fraction of medical waste. Plastic items commonly found in medical waste include disposable instruments, syringes, petri dishes, plasticized paperware, cutlery, plastic containers, packaging, bedpans, urine bags, respiratory devices, and dialysis equipment. The high chlorine content of these materials will increase the production of HCl. Baseline HCl emissions from MWCs burning municipal waste may range from 100 to 1000 ppm. Co-firing large amounts of medical waste may increase baseline HCl emissions.

Dioxin emissions are a second major concern. Medical waste incinerators have been found to emit dioxins at about the same levels as typical municipal waste incinerators (100-2000 ng/dscm) (4). Although dioxin formation is more strongly correlated with system design and operation, dioxin emissions may also be affected to some extent by waste characteristics. Medical waste has a number of characteristics that could lead to dioxin formation. A significant amount of aromatic compounds are present that may serve as dioxin precursors. These compounds include cytotoxins, laboratory wastes such as xylene, and components of various packing materials. Medical waste also contains a high concentration of chlorinated plastics and is highly non-homogeneous. Some components of the waste have high moisture contents. These materials may create localized low temperature zones within the waste bed, resulting in fuel-rich pockets that may escape complete destruction in the furnace.

Metals emissions are a third major concern. Medical waste contains such toxic metals as mercury, cadmium, and arsenic. These metals are volatile and may vaporize in the incinerator. The metal vapors would then be carried away in the exhaust gases. As the exhaust gases cool, the metals would condense to form small particles which are difficult to capture. A large fraction of these particles would be emitted to the atmosphere. Mercury, cadmium, and arsenic have been detected in hospital waste incinerator exhausts (2, 5).

IMPORTANT CONSIDERATIONS

Beyond emissions concerns, a number of other factors must be considered when burning medical waste in MWCs. Transportation and hauling is an important factor when medical waste is incinerated off-site as compared to an on-site hospital incinerator. Incinerator operation characteristics such as temperature, and time at a given temperature are also important factors relating to pathogen and cytotoxic chemical destruction. Ash handling and disposal are important as well due to potential pathogen survival, radioactivity, and metals content of the ash.

Waste Transportation and Handling

The transportation of medical waste to municipal waste incinerators and the subsequent handling of the waste at the incinerator site are areas of significant concern. As mentioned earlier, a number of medical waste components are extremely dangerous and must not be allowed to enter the environment. Precautions must be taken to ensure that none of the waste materials escape during transportation. These precautions include placing the waste in sealed, secure containers and procedures to deal with accidental releases such as those that may occur during a traffic accident. A truck itself is not considered a suitable containment system; rather it is a transport mechanism only. All medical waste should be placed in rigid or semi-rigid leakproof containers before being loaded onto a truck (6).

Storage time and temperature are important considerations. Microbial growth and putrefaction rates increase with temperature, resulting in increased potency. In addition, unpleasant odors associated with decaying organic matter are produced. EPA recommends that storage times be kept as short as possible (6).

Once the waste is on site, transport containers must not be opened or breached until they are either in the combustor or in a secure environment. Medical waste cannot be preprocessed because of the resulting increased chance of exposure. All personnel handling medical waste should be aware of its nature and take proper precautions. However, if the integrity of the containers has not been compromised, it is unlikely that specialized clothing or respirators would be needed. All equipment used to handle waste containers should be sterilized periodically (6).

Operation

When medical waste is burned in municipal waste incinerators, several operational precautions must be observed. It is generally believed that temperatures above 1800°F (982°C) are required to destroy cytotoxic compounds (3). In addition, temperatures must be maintained at sufficient levels to ensure the complete destruction of all pathogens in the waste. However, one current study indicates that typical MSW operating conditions may be sufficient to accomplish this (7). One final requirement is that medical waste be rendered "unrecognizable." That is, one should not be able to recognize body parts and medical equipment in the residual ash. This essentially requires good burnout of all incoming waste types. This is challenging because some materials (hospital gowns) are treated with flame retardant for safety reasons during their normal use.

Ash Handling

The primary concern in handling the ash from an incinerator burning medical waste is to ensure that all infectious agents have been destroyed. In general, the conditions found in a municipal waste incinerator will be sufficient to destroy most pathogens. However, additional concerns exist with regard to the ability of some viruses to survive incineration. A generally accepted test that demonstrates the destruction of infectious agents has not yet been developed.

A second concern stems from the fact that radioactive materials may be present in medical waste. Radioactive materials may be concentrated in the ash, producing a material that cannot be placed in a municipal landfill. The radioactive nature of the waste must be carefully monitored to ensure that it does not exceed acceptable levels.

The metals present in medical waste are a third area of concern. Studies of the ash from hospital incinerators indicate that relatively large quantities of cadmium and lead are present (8). These metals may potentially leach from the ash into groundwater reservoirs. The metals content of MSW ash is currently an important issue. Therefore, the impact on the metals content of the ash produced by an MWC also burning medical waste presents a definite concern as well.

CURRENT PRACTICES

Medical waste is routinely fired in at least 11 MWC facilities, including two mass burn waterwall, one mass burn refractory, five modular starved air, and three modular excess air combustors. Medical waste comprises less than 5 percent by weight of the total feed for 10 of these systems and 50 percent by weight for 1 of the modular starved air units. Three additional MWC facilities—two mass burn refractory and one fluidized bed combustor—previously accepted medical waste but have ceased this practice. Design and operating characteristics of these 14 facilities are summarized in Table 1. The waste types, handling procedures, and reported problems are summarized in Table 2. A more detailed description of each of the facilities is outlined in the following sections.

MASS BURN WATERWALL MWCs

Facility 1 comprises three units, each with 375 tpd (340 tonnes/day) capacity. Facility 2 includes two units rated at 275 tpd (249 tonnes/day) each. Both plants are largely considered to be state-of-the-art combustors, with only minor differences in design and operation. Facility 1 reports firing 0.2 percent by weight medical waste and Facility 2 reports firing less than 1 percent. Both facilities have ram feeders. Feed rates are automatically controlled according to steam demand and excess oxygen levels. Five underfire air plenums supply 60–80 percent of the total combustion air. Three rows of overfire air jets supply the remaining portion of air to the primary combustor. Only the underfire air is preheated. Facility 2 uses

TABLE 1. DESIGN AND OPERATING PROCEDURES OF MWCs ACCEPTING MEDICAL WASTE

PLANT	# OF UNITS	UNIT CAPACITY (tpd)	UNIT CAPACITY (tonnes/day)	COMBUSTION TYPE	MEDICAL WASTE MASS INPUT (%, AVERAGE)	AIR POLLUTION CONTROL DEVICE
1	3	375	340	Mass burn waterwall	0.2	ESP
2	2	275	249	Mass burn waterwall	<1	SD/FF
3	4	250	227	Mass burn refractory	1	ESP
4	2	120	109	Mass burn refractory	<1	Water sprays
5	4	250	227	Mass burn refractory	0.4	Wet scrubber
6	2	50	45	Modular starved air	0-5	ESP
7	3	36	33	Modular starved air	≤2	Fabric filter
8	4	20	18	Modular starved air	<1	Fabric filter, wet scrubber, packed tower
9	3	90	82	Modular starved air	50	Dry injection/ ESP
10	3	38	34	Modular starved air	≤1	None
11	2	75	68	Modular excess air	2	ESP
12	2	25	23	Modular excess air	<1	ESP
13	3	38	34	Modular excess air	1	ESP
14	2	105-120	95-109	FBC	5	Venturi

TABLE 2. WASTE TYPES, HANDLING PROCEDURES, AND REPORTED PROBLEMS IN MWCs ACCEPTING MEDICAL WASTE

FACILITY		WASTE TYPE ACCEPTED											PROBLEMS EXPERIENCED					HANDLING
		REFUSE	CULTURES & STOCKS	PATHOLOGICAL	BLOOD AND BLOOD PRODUCTS	USED SHARPS	ANIMAL CARCASSES	ISOLATION WASTES	UNUSED SHARPS	RADIOACTIVE	CYTOTOXIC	HAZARDOUS	SHARPS FALL THROUGH	INCOMPLETE BURNOUT	TEMPERATURE CONTROL	CONTAMINATION	REDUCED THROUGHPUT	
MASS BURN WATERWALL	1	X	X		X	X		X	X				X					Hauler places in grapple
	2	X	X		X	X		X	X		X		X					Separate in pit
MASS BURN REFRACTORY	3	X		X	X			X			X					X		Hauler places on conveyor
	*4	X	X	X	X	X	X	X	X		X		X	X				Hauler places in crane
	*5	X	X	X	X	X	X	X	X		X		X	X	X	X	X	Separate in pit
MODULAR STARVED AIR	6	X	X		X	X		X	X		X							Hauler places in hopper
	7	X	X		X	X	X	X	X		X							Hauler places in hopper
	8	X	X			X		X	X		X		X					Hauler places in hopper
	10	X	X	X	X	X	X	X	X		X							Hauler places in loader
MODULAR EXCESS AIR	11	X	X			X		X	X								X	Separate in pit
	12	X	X	X	X	X	X	X	X		X			X	X		X	Hauler places in crane
FBC	*14	X	X	X	X	X		X	X		X							Shredded

*No longer accepting medical waste.

natural gas as auxiliary fuel to achieve temperatures of 1800°F (982°C) at the top of the furnace prior to waste feed during start-up and until all the waste is burned off the grate during shutdown. Facility 1 does not use auxiliary fuel. Facility 1 is equipped with three-field ESPs with typical operating inlet gas temperatures ranging from 375 to 505°F (190 to 263°C). Facility 2 is equipped with spray dryers and fabric filters.

Facility 1 receives medical waste from commercial haulers. A significantly higher tipping fee is charged for the medical waste than for municipal waste. No body parts, animal carcasses, or radioactive wastes are accepted. Sharps are accepted. The waste is delivered in lined boxes. The haulers manually load the boxes into a closed grapple which is unloaded directly to the charging hopper. The medical waste is mixed with municipal waste in the hopper in approximately equal amounts and incinerated immediately. The crane is cleaned after every load and the haulers are responsible for cleaning any spill from broken boxes. Coveralls, dust masks, and gloves are worn at all times by facility personnel. Facility personnel, as well as the haulers, receive training in the handling and operating procedures required by the facility.

Facility 2 also receives waste from commercial haulers. Prior to [illegible] of January 1989, 70 tons/month (63 tonnes/month) of medical waste we[illegible] delivered from out of state at a tipping fee of $300/ton ($331/tonne). [illegible]his contract was scheduled to expire at the end of the month with no plans for its renewal. The facility plans to continue receiving 15-20 tons/month (14-18 tonnes/month) of medical waste from within the county for $26/ton ($29/tonne), the same amount charged for municipal waste. Like Facility 1, no body parts, animal carcasses, cytotoxic wastes, or radioactive wastes are accepted. Sharps are accepted. Before the expiration of the out of state contract, the waste was delivered in sealed boxes, unloaded to a conveyor, and transported directly to the hopper. The in-county medical waste is delivered in red plastic bags and dumped into a separate area in the pit. An overhead crane is used to transport the red bags from the pit to the hopper. The crane sometimes penetrates the bags and plant personnel are presently in the process of requesting that all medical waste be delivered in sealed boxes. The medical waste is mixed with municipal waste in alternating layers in the hopper and incinerated immediately after delivery. The conveyor is cleaned after every load. The tipping floor and edges of the pit are decontaminated periodically and the crane is cleaned prior to maintenance. Surgical gloves, goggles, and respirators are available if any close contact or exposure to medical waste is required. Personnel training includes instruction from local medical professionals regarding infectious disease transmission, decontamination procedures, and precautions.

Both Facilities 1 and 2 report good burnout and routinely test the bottom ash for pathogens. The test results reportedly have always been negative. Neither facility reported any combustion problems or increased HCl or metals emission attributed to the medical waste. Reportedly, mixing the medical waste with the municipal waste prior to combustion eliminates any potential problems associated with the widely variable heat content of the medical waste. Both facilities did report that sharps occasionally fall through the grate system into the riddlings hoppers. Facility personnel are specifically instructed not to reach into this area with their hands during maintenance to avoid injury.

MASS BURN REFRACTORY MWCs

Three mass burn refractory facilities reportedly fire medical waste along with MSW. Facility 3 consists of four 250-tpd (227 tonnes/day) rectangular refractory wall combustors with rocking grates. The waste is gravity fed. Medical waste reportedly consists of 1 percent of the total waste feed. The feed rate is controlled automatically by the furnace exit temperature and manually adjusted based on waste characteristics and overall waste volume reduction. The furnace exit temperature ranges from 1400 to 1800°F (760 to 982°C). Combustion air is not preheated and no auxiliary fuel is used. There is no heat recovery. The air pollution control devices include a water quench chamber for temperature reduction and two-field ESPs for particulate control. The ESP inlet gas temperature is 500°F (260°C).

Commercial haulers deliver approximately 300 boxes of medical waste per day to Facility 3. No radioactive waste, body parts, animal carcasses, or sharps are accepted at the plant. The tipping fee for medical waste is more than twice the municipal waste fee. Haulers unload the boxes to a conveyor which transports the waste directly to the charging hopper, as opposed to the municipal waste which is dumped into a pit and loaded into the hopper with an overhead crane. Boxes occasionally fall off the conveyor exposing the contents, in which case the hauler is responsible for cleaning and decontaminating the area. Facility 3 does not provide facility personnel with any special training or protective clothing or equipment since the commercial haulers are totally responsible for waste handling. Facility 3 did not report any combustion problems or emission increases attributed to the medical waste. Reportedly, the medical waste helped to maintain temperatures in the combustor and good burnout is achieved. Neither the stack emissions nor the bottom ash have reportedly been tested for pathogens.

Prior to the passage of recent state medical waste regulations, Facility 4 fired less than 1 percent hospital waste in two rectangular refractory-wall combustors that utilize three rocking grate sections per combustor. Each unit has a firing capacity of 120 tpd (109 tonnes/day) of MSW. In light of these regulations, the facility is no longer accepting medical waste. In addition to burning MSW, the plant also burns skimmings from the sewage treatment plant. A total of 417 tons (378 tonnes) of sludge was reportedly burned in 1986. Waste is charged from a holding pit into a water-cooled hopper which feeds each combustor by gravity. The feed rate is controlled by varying the speed of the first (drying) grate section. The majority of the burning takes place on the second grate section, and burnout is completed on the third (finishing) grate. Bottom ash is discharged from the finishing grate to a wet quench. A drag chain conveyor transports the ash to a truck for disposal in a nearby landfill.

Underfire air is supplied by forced-draft fans (one per combustor). Siftings hoppers are located beneath the drying grate. Separate forced-draft fans (one per combustor) located adjacent to the underfire air fans supply overfire air. All adjustments in air flow are made manually based on visual observation of the burning bed and flame patterns. Grate speeds are also varied manually by controls on the side of the combustor. The speed of each grate section can be independently set and varied.

There are no auxiliary fuel burners in either combustor. The combustor is started up by establishing a bed of waste on the first grate section and igniting the waste by hand. Plant operators reported that during start-up it takes approximately 1 to 2 hours to achieve a temperature of 1400°F (760°C) in the combustion chamber. When this temperature is achieved, the overfire air is introduced and the furnace temperature is established at 1700 to 1800°F (927 to 982°C).

Combustion products leaving the active burning region flow through a three-pass wet baffle system which both cools the hot flue gases and reduces particulate matter (PM) emissions. After passing through the wet baffle system, flue gases from both combustion trains are combined in a short run of ducting to the stack. Water and PM collected in the baffle system flow to a concrete lagoon where the ash settles out from the water. Every 3 to 4 months this ash is dredged out and disposed of at a nearby landfill.

During the period in which Facility 4 was receiving medical waste, no radioactive waste was accepted. Pathological waste, animal carcasses from the humane society, and sharps were accepted. Commercial haulers delivered the waste in red plastic bags and sharps in rigid boxes. The haulers manually placed the medical waste into the crane bucket or carried the waste up in an elevator and dumped the waste directly into the charging chute. The medical waste was then immediately incinerated. Some medical waste believed to be from doctors' or dentists' offices was also mixed in with the municipal waste. No combustion problems were attributed to the widely variable heat content of the waste because of the small amount of medical waste received. Likewise, no increase in HCl or metals emissions was noted. Neither the stack emissions nor the bottom ash were ever tested for pathogens. The two main problems attributed to medical waste were sharps and animal carcasses. Needles fell through the grates into the riddlings hopper creating potential hazards for maintenance personnel. Also, needles were caught in the front end loader used to transport municipal waste and were discovered during maintenance. Animal carcasses also caused problems because of poor burnout and recognizability after the combustion process.

Prior to the expiration of a special permit, Facility 5 was firing 0.4 percent by weight hospital waste. However, due to operational problems attributed to the medical waste, the facility did not seek permit renewal and no longer accepts medical waste. Facility 5 consists of four split-flow rotary kiln mass burn refractory combustors each with a capacity of 250 tpd (227 tonnes/day). The waste is gravity fed. There are two grate sections, drying grates, and ignition grates prior to the rotary kiln. Preheated underfire air is supplied to the ignition grate section. No auxiliary fuel is used. The combustor gas temperature is maintained at 2000°F (1093°C) at the exit of the ignition chamber and 1800°F (982°C) at the exit of the mixing chamber. There is no heat recovery. The facility is equipped with a water spray chamber and a venturi scrubber.

During the period in which Facility 5 was receiving medical waste, no radioactive waste was accepted. Pathological waste and sharps were accepted. Commercial haulers were required to deliver medical waste between 6 and 8 AM so as not to interfere with municipal waste deliveries. A slightly higher tipping fee was charged for medical waste than for municipal waste. Medical waste was delivered in red plastic bags and rigid sharps containers. The waste was dumped into a separate area in the pit and loaded into the hopper by an overhead crane. Facility 5 did not provide facility personnel with any special training or protective equipment since the commercial haulers were responsible for all handling of medical waste. Many problems were attributed to the medical waste including increased temperature due to the high Btu content, reduced throughput, increased HCl emissions, sharps falling through the grates, and poor burnout. No pathogen testing was ever conducted on the stack emissions or the bottom ash.

MODULAR STARVED AIR SYSTEMS

There are five modular starved air facilities that fire medical waste along with MSW. Facility 6 includes two units with design capacities of 50 tpd (45 tonnes/day) each. Facility 7 has three units designed to fire 36 tpd (33 tonnes/day) each. For both facilities, the feed rate is manually adjusted based on the primary chamber temperature. Exit gas temperatures are typically 1200-1600°F (649-871°C) from the primary chamber and 1800-2000°F (982-1093°C) from the secondary (upper) chamber. Natural gas is used at Facility 6 for start-up. The units are required to achieve 700°F (371°C) in the upper chamber prior to initiating waste feed. Auxiliary fuel is not used for shutdowns. Facility 7 has auxiliary oil burners available, although they are not used for start-up or shutdown. Both Facilities 6 and 7 have heat recovery and generate 25,000 lb/hr (11,340 kg/hr) and 20,000 lb/hr (9072 kg/hr) of steam, respectively, per unit. Facility 6 is equipped with a two-field ESP which operates at an inlet gas temperature of 526°F (274°C). Facility 7 is equipped with a fabric filter.

A commercial hauler delivers 5 tpd of medical waste to Facility 6 in heavy cardboard boxes with plastic liners and rigid leakproof plastic sharps containers. No radioactive waste, body parts, or sharps are accepted. A higher tipping fee is charged for the medical waste than for the municipal waste. The boxes are first unloaded to the tipping floor in an area separate from the municipal waste and then manually loaded into hoppers where it is mixed with municipal waste. Since all containers are leakproof, no routine decontamination procedures are followed; however, in case of a spill, any exposed area is decontaminated immediately. Facility personnel receive special training and are provided protective clothing and equipment for medical waste handling. Facility 6 does not report any combustion problems or increased HCl or metals emissions attributed to the medical waste. Good burnout is reportedly achieved with no recognizability of the ash. Neither the stack emissions nor the bottom ash is tested for pathogens. Reportedly, mixing the medical waste with the municipal waste prior to combustion eliminates any potential problems associated with the widely variable heat content of the medical waste.

Commercial haulers deliver 1 to 2 tons per week (0.9 to 1.8 tonnes/week) of medical waste to Facility 7 in plastic bags and boxes. A slightly higher tipping fee is charged for the medical waste. No radioactive waste or body parts are accepted. The haulers manually place the medical waste in the hoppers and it is incinerated immediately. Facility personnel are not provided any special training, clothing, or equipment since the commercial haulers are totally responsible for medical waste handling. No combustion problems or increased HCl or metals emissions were attributed to the medical waste. Good burnout is reportedly achieved. The facility does not test the stack emissions or the bottom ash for pathogens.

Facility 8 consists of four units rated at 20 tpd (18 tonnes/day) each and reportedly fires less than 1 percent by weight medical waste. A ram waste feed system is used. There is no air preheat. Natural gas is used during start-up to attain a temperature of 1500°F (816°C) measured at the secondary chamber outlet before waste feeding is initiated. Natural gas is also used during shutdown, although no specific temperature is required. The gas temperature is maintained at 1750 to 1850°F (954 to 1010°C) at the exit of the primary chamber and 1800 to 2000°F (982 to 1093°C) at the exit of the secondary chamber. Steam is produced in each unit at a rate of 7000 lb/hr (3175 kg/hr). The air pollution control devices include a fabric filter, a wet scrubber, and a packed tower.

Facility 8 receives approximately 20 boxes per day of medical waste. No radioactive waste, body parts, carcasses, or liquids are accepted. Commercial haulers manually place the boxes into the hopper separate from the municipal waste. The boxes are not allowed to touch the tipping floor. The hopper is checked daily for spills or leaks from the boxes and decontaminated if necessary. Facility personnel routinely wear respirators while on the tipping floor. No combustion problems or increased HCl or metals emissions were attributed to the medical waste. An increase in operating temperature was reported for the medical waste as compared to the municipal waste but was not viewed as a problem. Due to the small amount of medical waste being incinerated, the combustion control system responded adequately to the increased temperature. Good burnout is reportedly achieved. No pathogen testing has been conducted. Reportedly, needles occasionally fall through the grate system to the riddlings hopper, but no injuries have ever resulted.

Facility 9, consisting of three units rated at 90 tpd (82 tonnes/day) each, fires 50 percent by weight medical waste and utilizes dry sorbent (sodium bicarbonate) injection with an ESP for air pollution control. All other design and operating characteristics have been declared confidential business information by the facility owner/operator.

Facility 10 consists of three units each with a 38-tpd (34 tonnes/day) capacity and reportedly fires less than 1 percent by weight medical waste. A ram waste feed system is used. There is no air preheat. Natural gas is used for start-up so that 1800°F (982°C) is achieved in the secondary chamber prior to waste feed. The gas temperature is maintained at 1600-1800°F (871-982°C) at the exit of the primary chamber and 1800-2000°F (982-1093°C) at the exit of the secondary chamber. Steam is produced at a rate of 10,000 lb/hr (4536 kg/hr). Facility 10 is not equipped with an air pollution control device.

Facility 10 was receiving autoclaved medical waste in red plastic bags mixed in with municipal waste since, according to state regulations, autoclaved medical waste can be treated exactly like municipal waste. This was not acceptable to the facility personnel because of safety concerns and because they had no way of knowing if the waste had in fact been autoclaved. Therefore, the facility implemented their own medical waste handling procedures. The autoclaved medical waste must be delivered separately from municipal waste. The hauler is required to manually place the bags into a front end loader whose operator is required to wear a respirator and protective coveralls. The waste is unloaded to the hopper where it is mixed with municipal waste and incinerated immediately. The coveralls are incinerated and the front end loader is steam cleaned after every load. No sharps or radioactive wastes are accepted and an extremely high tipping fee is charged. Since implementing these changes, no medical waste has been delivered to the facility and no operating information is available. The facility does not plan to implement any pathogen testing since only autoclaved medical waste is accepted.

MODULAR EXCESS AIR SYSTEMS

Three modular excess air facilities reportedly fire medical waste along with MSW. Facility 11 has two rotating annular hearths with capacities of 75 tpd (68 tonnes/day) each. The hearths intermittently rotate on a horizontal plane about a vertical shaft. The waste is gravity fed. Automatic pokers stoke the fuel bed at regular intervals when the hearth is at rest. The combustion chamber temperature is maintained at 1850°F (1010°C). The combustion gases leave the rotary annular hearth and are tangentially fired into a cyclonic secondary chamber which serves the dual purposes of providing initial particulate matter separation and a high degree of mixing. The secondary chamber exit gas temperature is maintained at 1800°F (982°C). The ash is discharged from the hearth by a fixed plow into a water quench tank. Facility 11 has waste heat boilers that generate 16,000 lb/hr (7258 kg/hr) of steam per unit. Oil is used as auxiliary fuel during start-up in order to achieve a boiler exit temperature of 320°F (160°C) prior to initiating waste feed. Auxiliary fuel is reportedly not used for shutdown although a temperature of 700°F (371°C) is maintained in the secondary chamber until all the waste is burned off the grate. Facility 11 is equipped with two-field ESPs which operate at an inlet gas temperature of 550-650°F (288-342°C).

Commercial haulers deliver boxed medical waste from a local hospital to Facility 11. The hospital incinerator is used for disposing of pathological waste, cytotoxic chemicals, and fluid-filled containers. All other medical wastes, except radioactive and hazardous wastes, are delivered to Facility 11. Sharps are sterilized prior to delivery. A higher tipping fee is not currently charged for the medical waste but is being considered. The boxed medical waste is dumped into a separate area of the pit and loaded into the hopper by an overhead crane which occasionally penetrates the boxes. The medical waste is incinerated immediately in the presence of the delivery personnel who must verify that all of the medical waste was in fact incinerated. Occasionally, if temperatures become too high, operators will mix the medical waste with municipal waste. Throughput is reduced considerably to ensure good burnout. No increase in HCl or metals emissions is attributed to the medical waste. No pathogen testing has been conducted. Due to the small "pin hole" design of the grate system, there are no reported problems of sharps falling through into riddlings hoppers.

Facility 12 consists of two 25-tpd (23 tonnes/day) modular excess air combustors with waste heat boilers and ESP controls. The plant has been operating commercially since May 1985, and the normal operating schedule is 24 hours per day, 5 days per week. Facility 12 fires less than 1 percent medical waste as well as 8 percent sewage sludge. A feed pump transfers the sludge to the combustor charging hopper where it is metered into the MSW feed. The waste is fed by a charging ram and is moved through the unit by a series of pneumatically driven pokers. The combustor operates with nearly 115 percent excess air and combustion takes place on stepped hearths in the primary combustion chamber, where the temperature reportedly averages 1700°F (927°C). The flue gas flows to a post-combustion chamber where average temperatures are reportedly near 1900°F (1038°C). There is no air injection in the secondary chamber; combustion air control is achieved by an induced draft (ID) fan. The flue gases flow from the combustor through the firetube boilers and ESPs, which are designed to operate at 450°F (232°C). The ESP plates are constructed of Corten steel to reduce corrosion effects. Each ESP has one field. The units include oil burners which are used during start-up and shutdown conditions. The ash is co-disposed of in a lined landfill with leachate collection.

Commercial haulers deliver red plastic bags and boxes of medical waste to Facility 12. Radioactive waste is not accepted. There is not a higher tipping fee for medical waste. The haulers manually place the medical waste into the crane, which is unloaded into hoppers and the medical waste is incinerated immediately. Facility personnel are not provided any special training, protective clothing, or equipment since the commercial haulers are totally responsible for all medical waste handling. The medical waste is not mixed with municipal waste. Resulting higher temperatures are controlled by reducing throughput. Reportedly, animal carcasses are occasionally recognizable in the ash. The ash has been tested for pathogens, although not routinely, with negative results. No increase in HCl or metals emissions is attributed to medical waste.

Facility 13 reportedly fires 1 percent medical waste in three 38-tpd (34 tonnes/day) units each equipped with a ram feed system. There is no air preheat. The average gas temperatures exiting the primary and secondary chambers are 2000°F (1093°C) and 1500°F (815°C), respectively. Each boiler produces steam at a rate of 7100 lb/hr (3220 kg/hr). Natural gas is used as auxiliary fuel for start-up to attain a temperature of 400°F (204°C) at the top of the primary chamber prior to initiating waste feed. Auxiliary fuel is also used for shutdown although there is no requirement to maintain a specific temperature until all of the waste is burned off the grate. Facility 13 is equipped with a two-field ESP which operates at an inlet gas temperature of 450°F (232°C).

Reportedly, Facility 13 receives medical waste mixed in with the municipal waste; the medical waste is believed to be generated by doctors' and dentists' offices. The facility is currently retrofitting one of their three units to burn only medical waste. Facility personnel were not willing to discuss any of the details of the retrofit or medical waste handling and operating procedures.

FLUIDIZED BED COMBUSTORS (FBCs)

Facility 14 combusts a mixture of sewage sludge and refuse derived fuel (RDF) in two FBCs. The units were primarily designed as sewage sludge incinerators that use RDF as a cheap auxiliary fuel. Facility 14 consists of two identical bubbling bed FBCs with individual waste heat boilers. The combustors are each 45 ft (14 m) high and the reactor vessels have an inside diameter of 34 ft (10 m) at the freeboard. Each combustor is capable of firing 120 tons/day (109 tonnes/day) of fluff-RDF and 345 tons/day (313 tonnes/day) of sewage sludge (18 percent solids); thus, RDF represents approximately 26 percent of the total waste input at full load. RDF is produced at Facility 14 8 hours/day, 5 days/week. Input waste to the processing plant consists of approximately 60 percent residential, 25 percent commercial, 10 percent industrial, and 5 percent medical waste. RDF is pneumatically injected into the bed through four ports which extend approximately 5 feet (1.5 m) from the reactor wall about 18 in (0.46 m) above the gas distribution plate. The feed ports are angled downward at the ends to introduce the RDF low in the bed. Sewage sludge is pumped through a nozzle which penetrates the top of the reactor vessel and extends approximately 12 ft (3.7 m) below the roof. Each 160 x 10^6 Btu/hr (169 x 10^9 J/hr) boiler can provide 49,000 lb/hr (22,226 kg/hr) of steam, and all steam is used on site. Wood chips are fired as a supplemental fuel whenever sufficient heat input cannot be derived from RDF. Usually, wood is required only when all of the RDF feeding tubes plug simultaneously or when the RDF supply is occasionally exhausted at the end of the weekend. Wood chips account for about 7.5 percent of the facility heat input on an annual basis. Oil is fired for cold start-ups and as a second supplemental fuel; the boilers can be fired at up to 100 percent load using oil. The sewage sludge provides less heat input than is required to dry it and is, therefore, not considered a fuel by the facility. At typical operating conditions, the FBC has a bed temperature of about 1475°F (802°C) and a freeboard (furnace exit gas temperature, or FEGT) of about 1650°F (899°C). Typically all of the combustion air is supplied as underfire air. Although the FBCs are configured for overfire air injection, it is seldom used. The flue gas has 5 to 7 seconds residence time between the combustor and the boiler entrance, and the gas temperature at the entrance to the first convective section is about 1400°F (760°C). Each of the units is equipped with a venturi scrubber, a quench tower, and a demister.

Facility 14 is no longer accepting medical waste because of the potential hazards associated with shredding in RDF processing. Facility 14 assigned one employee to work with the area hospitals to ensure that only general refuse and no medical waste was sent to the MWC facility.

CONCLUSIONS

The 14 municipal waste combustors that have previously accepted or are currently accepting medical waste include a variety of combustor design types. The amount of medical waste burned in these units ranges from less than 1 percent by weight to 50 percent. Some of the major concerns regarding medical waste incineration in MWCs include waste handling procedures prior to combustion, identification of appropriate waste types to be burned in MWCs, and evaluation of potential problems associated with residue handling, including ash and effluent streams. Based on the preliminary findings, it appears that sharps pose a significant hazard in MWCs with grate systems. Sharps in the ash represent potential health hazards to facility personnel or landfill operators coming in contact with the ash. Animal carcasses also represent a significant problem with poor burnout resulting in recognizability of the ash. Prior to incineration, attention must be given to the medical waste handling procedures to prevent penetration of containers and exposure of containerized wastes. For example, it is recommended that medical waste not be processed into RDF. There are very limited data available on the emission impacts associated with combustion of medical waste in MWCs. More research is needed to fully evaluate the potential impacts of burning medical waste on the emissions of acid gas, dioxin, and metals. Also, further studies and field tests are needed to determine the design and operating requirements for complete destruction of hazardous solvents, cytotoxic chemicals, and pathogens.

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FLUE GAS CLEANING SYSTEM OF PADUA INCINERATOR

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ABSTRACT

The municipal solid waste incinerator of the city of Padua, is equipped with a post combustion chamber that guarantees the following operative conditions:

- free oxygen percentage in the wet fumes	> 6% in volume
- medium speed of gases in entrance section of the chamber	> 10 m/s
- contact time	> 2 seconds
- temperature	> 950° C
- combustion efficiency	> 99,9 %

This chamber is followed by an heat recovery system with production of electric current.

Dry injection of hydrated lime in two parallel reactors for binding of acid bases is followed by two fields electrostatic precipitator.

For a period over ten months analisys of emission have been made. In the paper are reported the results of this analytical campaign for : particulates, HCl, HF, SO_2, NOx, Pb, Cd, Hg, PCDD, PCDF, IPA, PCN.

INTRODUCTION

The waste incineration plant of the city of Padua treats 130 tons per day of municipal solid waste and 15 tons per day of hospital waste. Hospital wastes come from almost all sanitary facilities existing in Veneto Region.

Veneto is a region located in the north-east of Italy with Venice as capital. This region has about 4 millions inhabitants.
Up to five years ago, each hospital had its own incinerator for hospital waste, but 4 years ago, in 1984, a law fixed new operating conditions that had to be met by incinerators:
- post combustion chamber;
- capacity over 100 tons/day;
- high combustion efficiency.

This regulation, which came into force at the end of 1986, caused the close down of about 50 incinerators of high and small dimensions. Only 4 incinerators do respect that rule and may continue to work.

For this reason, hospital wastes had to be directed to the two existing municipal waste incinerators of Padua and Schio. Hospital wastes are not dumped in the receiving bunker as municipal waste but, packed in a polietilene bag introduced in a cardboard box or in a polietilene drum, are directly put into the feed shaft with an automatic system.

PLANT GENERAL DESCRIPTION

Combustion chamber is composed by :
- a feed shaft in which municipal waste are feeded with a traveling crane and hospital waste directly introduced;
- a oleodynamic operating stoker;
- a moving grid with oleodynamic control.

Combustion air introduced in the chamber is composed by primary air blown undergrid to help combustion of carbonium burning on the grid, and secondary air introduced above the grid for combustion of volatile material burning over the grid.

Primary and secondary air of combustion may be conveniently adjusted and their distribution system is designed to create a good turbolence to realize an effective mixing between cumburent and combustible to maintain a more uniform heat distribution all over the chamber.

POST-COMBUSTION CHAMBER

The flue gas, coming from combustion chamber, passes in the post-combustion chamber that, according to italian law, must guarantee the following operating conditions :

- O_2 free in wet fumes (measured downstream in the chamber):>6% in volume,
- average gases velocity in chamber entrance section : > 10 m/s,
- contact time : > 2 s.,
- temperature : > 950° C.

This chamber is provided with a burner that is automatically started as temperature of fumes goes under 950° C. In Padua incinerator temperature is normally around 1050-1100° C without the necessity of using the burner.

HEAT RECOVERY

Thermoelectric group consists of a vertical superheater working mainly by radiation. Vapour generator is a tube nest type, vertically disposed, operating by convection. Produced vapour is sended to a turbine to produce electric energy.

DRY SCRUBBERS

Flue gases coming from air mixer enter in the two reactors designed for dry absorption of acid gases. Dry scrubbers are composed of two parallel units. Reactors have a double step reaction, so that an higher efficiency of absorption of acid gases is obtained. Downstream each reactor is installed a cyclone for partial recycle of reagent. The gas control is obtained by injection of reagent through nozzles that realize a chemi-absorption of acid gases on line, giving as products salts as $CaCl_2$, CaF_2, $CaSO_3$, $CaSO_4$.

These compounds have to be captured by an electrostatic preciptator together with dust coming from combustion of waste and residual lime.

Absorption efficiency is depending on :

- specific surface of the dry absorbent per cubic meter of gas;
- concentration of polluting acid gases;
- contact time;
- homogeneity of mixing;
- temperature of reaction;
- water content of gases.

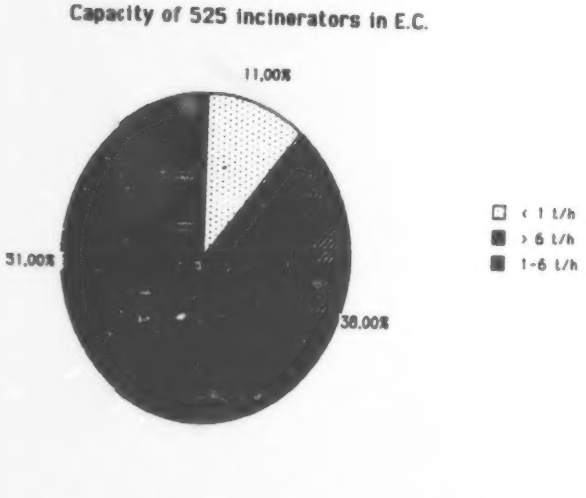
ESP

Solid particles - dust from combustion chamber, salts produced by reaction between acid gases and lime, and unreacted lime, are carried by the flue gas to an electrostatic precipitator. Fly ash, classified as a toxic waste by the italian law for the high content in Lead and Cadmium, is sended to a stabilization-solidification plant in Modena (Italy).

INCINERATORS IN EUROPEAN COMMUNITY

In European Community, as resulted by a specific research carried out by a Committee of European Community in 1988, are operating 525 incinerators. As shown in Fig. 1, 38% of them have a capacity higher than 6 tons per hour, 51% are in the range from 1 to 6 tons per hour, and 11% are under 1 ton per hour.

Fig. 1



In Fig. 2 it is shown that 88% of municipal solid waste is incinerated in large plants and only 12% in medium or small plants.

Fig. 2

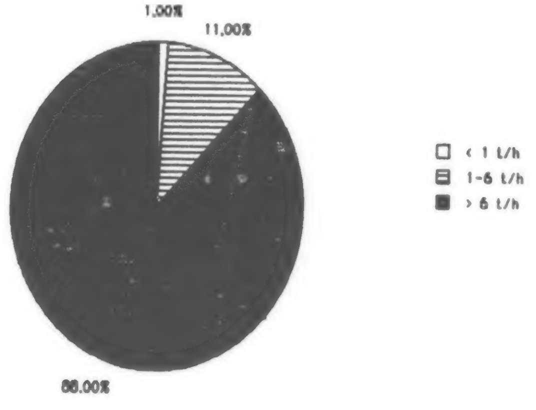
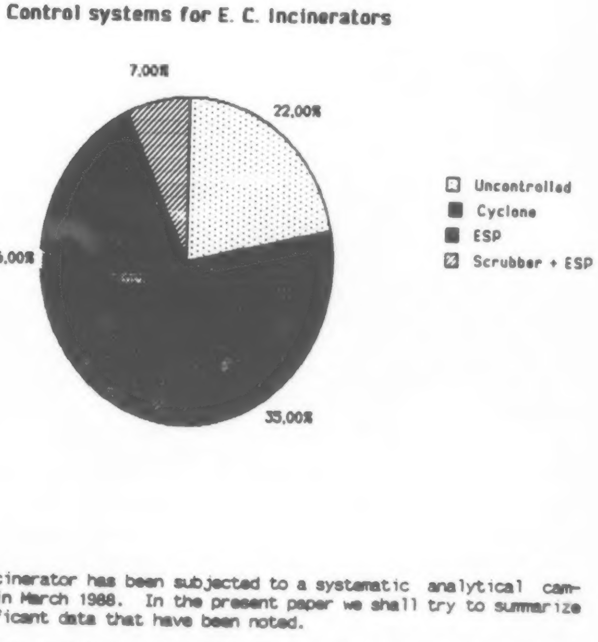

MSW incinerated compared to capacity of incinerators

Fig. 3 shows that 22% of European Community incinerators have no control systems at all, 35% are provided with mechanical devices only (cyclones or multiple cyclones), 36% with just an electrostatic precipitator and only 7% of working plants have both a gas treatment device (wet or dry scrubber)and an ESP or a baghouse.

Fig. 3


Control systems for E. C. Incinerators

Padua incinerator has been subjected to a systematic analytical campaign started in March 1988. In the present paper we shall try to summarize the most significant data that have been noted.

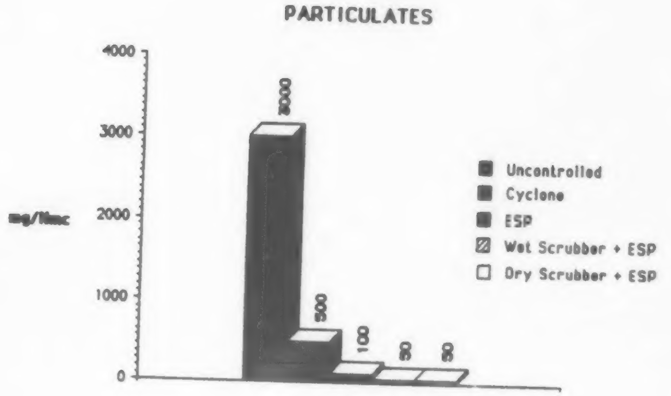
Measurements have been conducted by University of Verona up to August 1988 and by "Fondazione Clinica del Lavoro" of Padua from September to December 1988.

Our Company has an Environmental Department that co-operated with these two institutes during all the campaign.

PARTICULATES

In Fig. 4 are reported typical values for particulates in uncontrolled and controlled flue gases of European incinerators. Different kinds of control devices are considerated : Cyclones, Electrostatic Precipitators (ESP); Wet Scrubber followed by and ESP; Dry Scrubber followed by an ESP.

Fig. 4



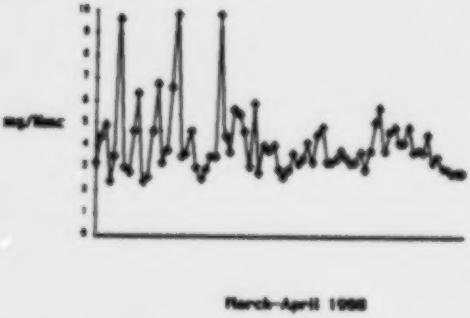
Emission limit proposed by European Community is 50 mg/Nm³ (referred to dry gas with 9% of CO_2) for incinerator with capacity higher than 5 tons per hour (100 is the limit for capacity lower than 5).

Different efficiencies of control systems considered are clearly pointed out.

In Fig.5 are reported results of continous analisys (Hartmann & Braun beta ray absorption system) at the stack for particulates. Values are in the range 2-10 mg/Nm^3.

Fig.5

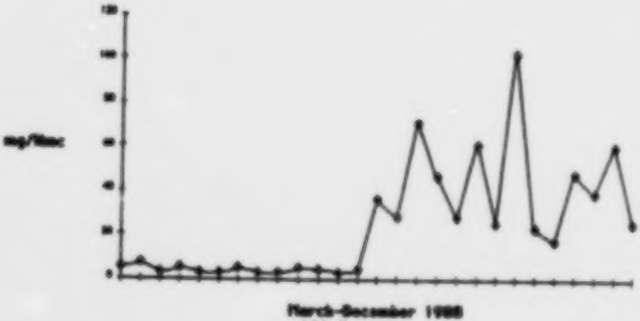
PARTICULATES



In Fig. 6 are reported other data surveyed by gravimetric analysis. Values are in the range 3-102 mg/Nm^3.

Fig. 6

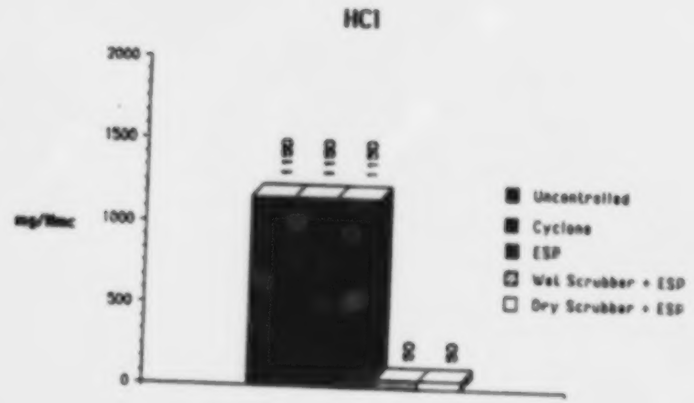
PARTICULATES



HYDROCHLORIC ACID

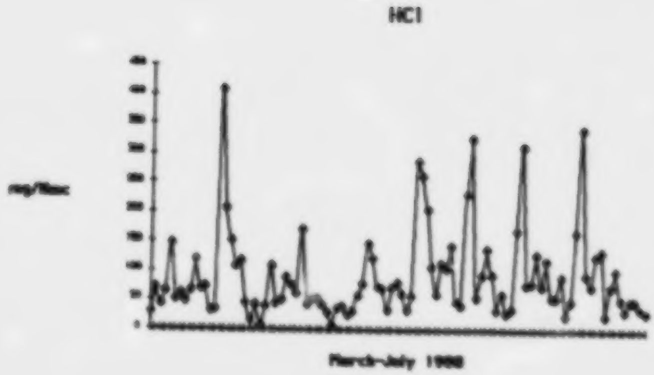
In Fig. 7 are reported typical values for HCl in uncontrolled and controlled flue gases of European incinerators. Different efficiences of control systems considered are clearly pointed out. It is clear that the use of a scrubber is a must. Emission Limit proposed by European Community is 50 mg/Nm^3 (referred to 9% of CO_2) as a monthly average. Daily average should not exceed for more than 30% this limit value.

Fig. 7



In Fig. 8 are reported data surveyed for Padua incinerator in the period March-July 1988.

Fig. 8



Values are in the range 6-2242 mg/Nm^3. It can be noted that data are very spread. This is depending on the great variability of concentrations of HCl in untreated gas due to the non-homogeneity of waste that is incinerated, particularly in our case with 15% of hospital waste characterized by an high content of Chlorine.

Being impossible to know the current concentration of HCl, the quantity of lime has been fixed to a certain value that cannot be the ideal for all circumstance. This is the reason for the different values noted.

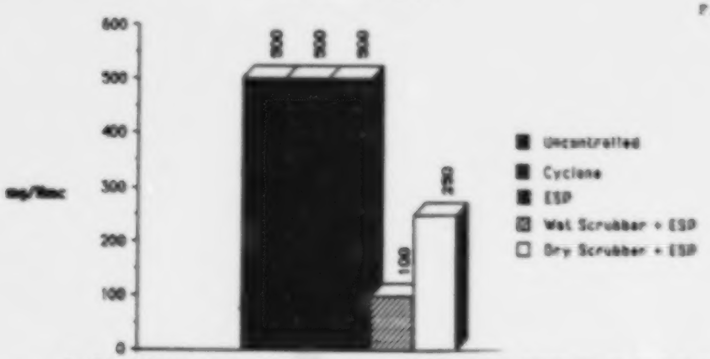
The only way to overcome the situation is to provide continous monitoring of HCl, as proposed by European Community for new plants.

SULFUR DIOXIDE

In Fig. 9 are reported typical values for SO_2 in uncontrolled and controlled flue gases of European incinerators. A wet scrubber is a more efficient device than a dry one. Emission limit proposed by European Community is 300 mg/Nm³ (referred to 9% of CO_2).

SULFUR OXIDES

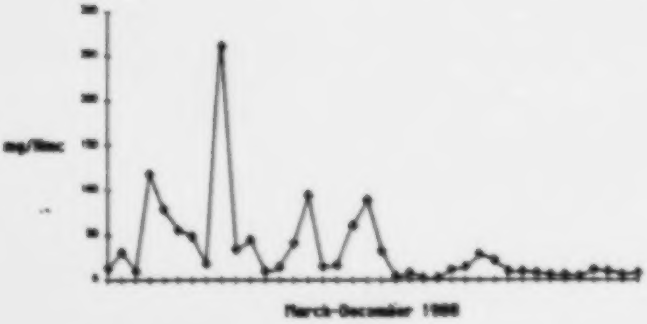
Fig. 9



In Fig. 10 are reported data surveyed for Padua incinerator in the period March-december 1988. Values are in the range 3-260 mg/Nm³.

SO2

Fig. 10



NITROGEN OXIDES

In Fig. 11 are reported typical values for NOx in uncontrolled and controlled flue gases of European incinerators. As aspected, control efficiencies for NOx are very low in all cases. No emission limits are fixed.

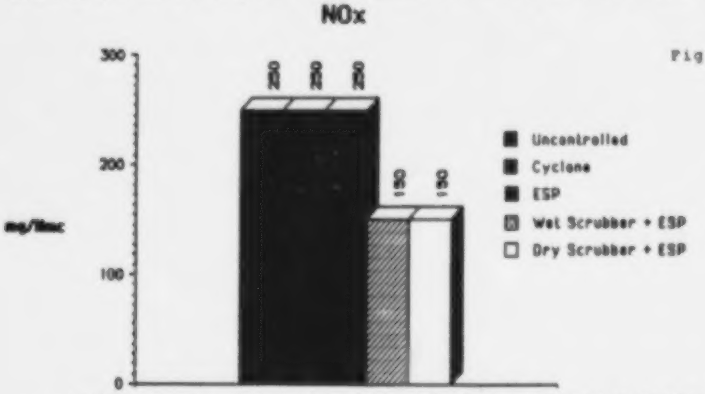


Fig. 11

In Fig. 12 are reported data surveyed for Padua incinerator in the period March-December 1988. Values are in the range 1-83 mn/Nm^3.

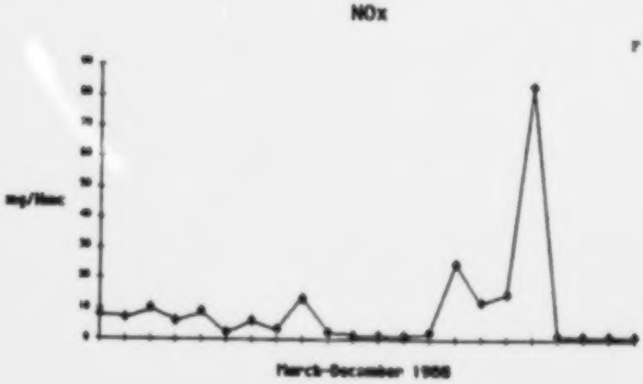


Fig. 12

LEAD

In Fig. 13 are reported typical values for Pb in uncontrolled and controlled flue gases of European incinerators. Emission limit proposed by European Community is 5 mg/Nm^3 (referred to 9% of CO_2), comprehensive of Cr, Cu, and Mn. Efficiency of control devices is closely dependent on their ability in controlling particulates.

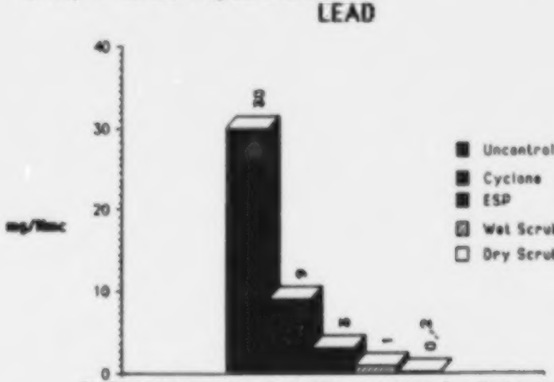


Fig. 13

In Fig. 14 are reported data surveyed for Padua incinerator in the period March-December 1988. Values in the range 0,028-1,097 mg/Nm^3.

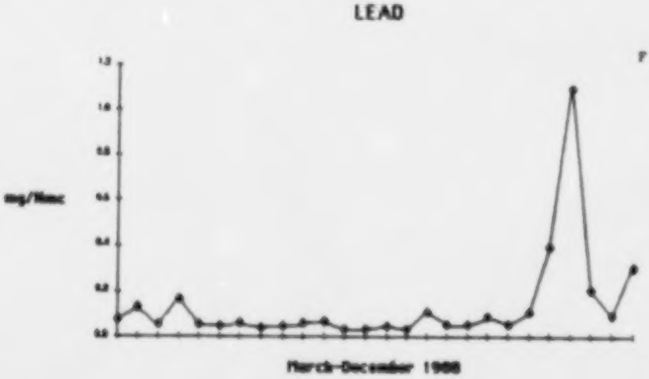


Fig. 14

CADMIUM

In Fig.15 are reported typical values for Cd in uncontrolled and controlled flues gases of European incinerators. Emission limit proposed by European Community is 0.1 mg/Nm³ (referred to 9% of CO_2).Efficiency of control devices is closely dependent on their ability in controlling particulates.

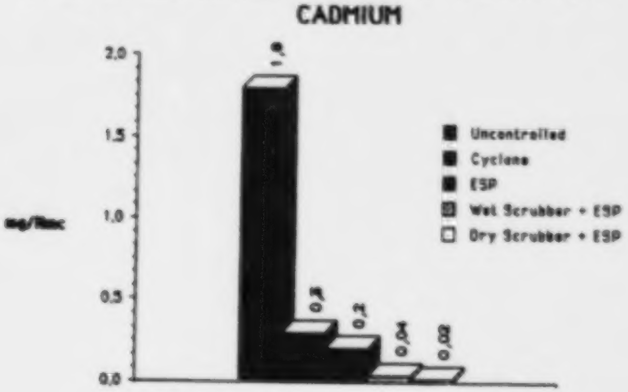


Fig. 15

In Fig. 16 are reported data surveyed for Padua incinerator in the period March-December 1988. Values are in the range 0,001-0,061 mg/Nm³.

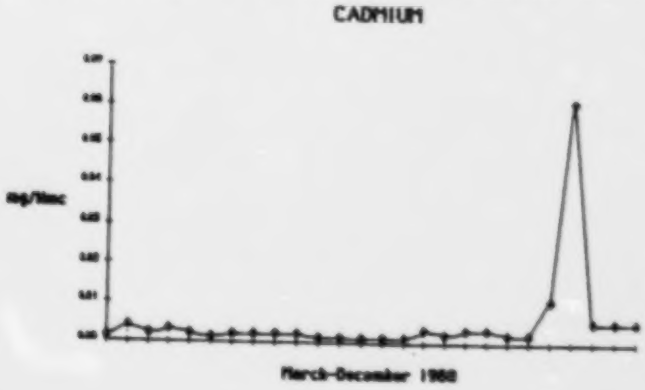


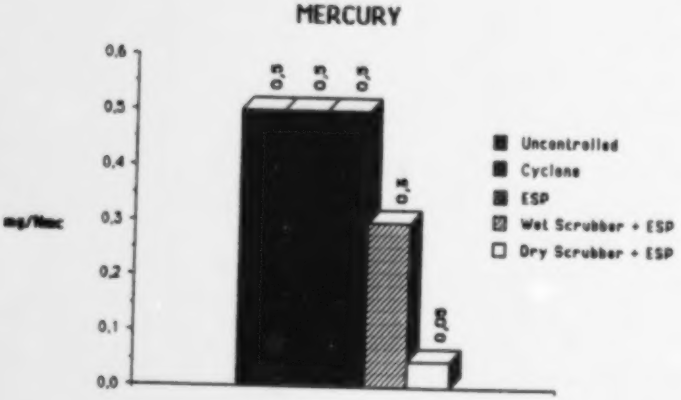
Fig. 16

MERCURY

In Fig. 17 are reported typical values for Hg in uncontrolled and controlled flue gases of European incinerators. Emission limit proposed by European Community is 0.1 mg/Nm³ (referred to 9% of CO_2).

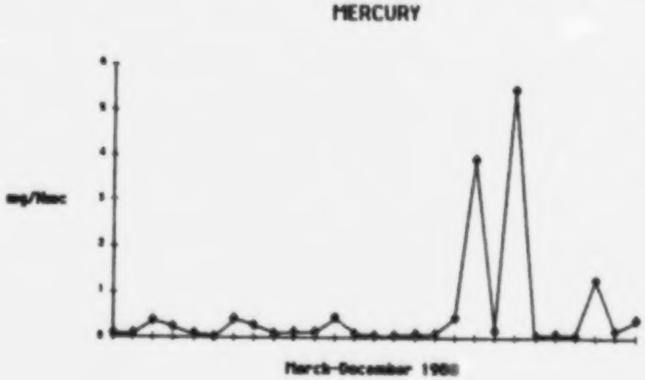
This metal is present in vapour fase in emission and for this reason is hardly controlled by any kind of device if temperatures are not lowered under 160 C. Above this temperature Mercury concentration cannot be reduced, so that it is the same outside combustion chamber and at the stack.

Fig. 17



In Fig. 18 are reported data surveyed for Padua incinerator in the period March-December 1988.

Fig. 18



Values are in the range 0,011-5,483 mg/Nm3^3. The reason is closely dependent on the fact that temperature of flue gases at the stack is about 200 C. We are studying to improve the capacity of heat exchanger to cool down flue gases in the dry scrubber.

TRACE ORGANICS EMISSIONS

In the period August-December 1988 "Fondazione Clinica del Lavoro" has analyzed trace organics emissions in flue gases at the stack (Table 1).

PCDD and PCDF resulted always under detection limit (2 ng/Nm^3).

The undoubted identification of PCDD, PCDF, PCB and PCN peaks in the corresponding gas chromatograms is obtained by checking simultaneous respect of retention time and ratio of areas of the three peaks of ionic chromatograms corresponding to the three more significant mass ions of isotopic cluster. For instance, absolute certainty that a gas-chromatographic peak, with the retention time of 3,4,7,8-TCDD, is really corresponding to this substance is obtained verifying that peaks of ions 320,322 and 324 have the same retention time and that their areas ratio is exactly the same of isotopic cluster: 76/100/49.

TABLE 1. EMISSIONS OF TRACE ORGANICS (ng/Nm^3)
August - December 1988

	Average	Std deviation
PAH		
naphthlene	627	549
acenphthene	29	44
biphenyl	687	1253
fluorene	32	32
phenanthrene	251	178
anthracene	29	28
fluoranthene	98	97
pyrene	90	64
benz(a)anthracene	36	36
chrysene	43	70
benzofluoranthenes	7.5	16
benzo(e)pyrene	1.7	4
benzo(a)pyrene	1.7	4
indeno(1,2,3-cd)pyrene	< 1	-
dibenz(a,h)anthracene	6	14
benzo(ghi)perylene	< 1	-

	Average	Std deviation
PCB		
dichloro	338	814
trichloro	1819	2683
tetrachloro	1860	2486
pentachloro	582	623
hexachloro	626	1098
heptachloro	398	564
octachloro	< 8	-
PCN		
dichloro	4684	11473
trichloro	264	582
tetrachloro	2444	3735
pentachloro	< 8	-
hexachloro	< 8	-
heptachloro	< 8	-
octachloro	< 8	-
PCDD		
dichloro	< 2	-
trichloro	< 2	-
tetrachloro	< 2	-
2,3,7,8-tetrachloro	< 2	-
pentachloro	< 2	-
hexachloro	< 2	-
heptachlor	< 2	-
octachloro	< 2	-
PCDF		
dichloro	2	-
trichloro	< 2	-
tetrachloro	< 2	-
pentachloro	< 2	-
hexachloro	< 2	-
heptachlor	< 2	-
octachloro	< 2	-

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

DESIGN CONSIDERATIONS AND SELECTION OF AN EMISSION CONTROL SYSTEM OPERATING AT LOW TEMPERATURES FOR A MSW COMBUSTION FACILITY

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ABSTRACT

The design of acid gas and particulate emission control systems for MSW plants is being made more difficult by the increasingly stringent emission regulations, coupled with boiler systems operating at low flue gas temperatures to increase station power efficiency (resulting in low inlet temperatures to the spray dryer). With very little empirical field data available, Westinghouse had to carefully evaluate the design options of the York County Resource Recovery Facility. This paper discusses the requirements, options, theoretical and practical considerations and describes the selected system.

The facility will have three separate combustion/boiler trains, each rated at 448 tons per day of MSW. Each train will have its own emission control system to remove acid gases and particulate matter. Start-up is projected for Summer, 1989.

INTRODUCTION

In November, 1985, Westinghouse was selected to design, construct, and operate a 1344 TPD resource recovery facility to be located in Manchester Township, York County, Pennsylvania. This plant is now under construction and will begin start-up operations in the Summer of 1989. The facility will generate approximately 35 megawatts (MW) of electricity and will export approximately 30 MW to Metropolitan Edison.

The York County Resource Recovery Facility (RRF) was designed to meet a comprehensive set of control criteria published by the Pennsylvania Department of Environmental Resources (PaDER). PaDER finalized the Best Available Technology (BAT) Criteria in August, 1986. The BAT Criteria required Municipal Waste Combustors (MWC) to install and operate with start-of-the-art air pollution control (APC) equipment. This equipment includes automatic combustion controls, acid gas scrubbers and high efficiency particulate matter removal equipment.

The BAT required the APC system to meet the following levels:

- HCl emissions - 30 ppm_{dv} @ 7% O_2 or 90% control
- SO_2 emission - 30 ppm_{dv} @ 7% O_2 or 70% control
- particulate emissions - 0.015 gr/dscf @ 7% O_2

AIR POLLUTION CONTROL STRATEGY

The York County facility will incorporate modern combustion controls and APC equipment. Each combustor/boiler train has individualized combustion controls and separate APC equipment. These systems are necessary to ensure that the facility meets PaDER's stringent emission requirements.

The facility is designed to extract as much heat as possible from the flue gas to improve the overall boiler efficiency. Each combustor/boiler consists of a water-cooled rotary combustor; radiant water well tubes, superheater, a convection tube bank and an economizer. Flue gas enters the radiant section in excess of 1800° F and leaves the economizer at approximately 350 to 400° F. The economizer has a bypass to maintain the gas temperatures equal to or greater than 350° F.

A number of regulatory agency engineers have recently written permit limits specifying maximum flue gas operating temperatures at either the spray dryer or baghouse outlets. These permit conditions limit operating temperatures to 290 to 325° F. The purpose of requiring specific maximum temperatures seems to be geared toward maximizing the efficiency for removing heavy metals (particularly mercury), dioxins, furans, and other organics. It is thought that many of these compounds are collected as particulate matter at temperatures less than 300° F; however, there is limited data to confirm this theory. In addition, acid gas removal efficiency, particularly SO_2 reduction, increases as flue gas approaches the saturation temperature.

In order to meet the stringent PaDER requirements at the designed operating temperatures, Westinghouse engineers decided to use a spray dryer and baghouse system as opposed to a spray dryer and electrostatic precipitator system. Spray dryer/baghouse systems have the ability to provide additional acid gas control as the flue gas passes through the dust cake that contains some unused sorbent (lime). The cake will help levelize the acid gas emissions particularly SO_2 emissions, thereby providing assurance that the system can continually meet the regulations.

KEY DESIGN CONSIDERATIONS

The key design considerations were basically three-fold:

- Operate the boiler so that the inlet flue gas temperature to spray dryer approaches 350° F to maximize boiler efficiency.
- Procure APC equipment that meets PaDER's requirements while operating at low inlet temperatures.
- Produce a dry product in the spray dryer and baghouse so that bags are not caked with damp calcium salts (mainly $CaCl_2$) or so that the collected fly ash moves freely through the hoppers and ash removal equipment.

The major challenge was to obtain an APC system that could perform reliably while operating under the above-mentioned design constraints; low inlet temperatures, low outlet temperatures and therefore a relatively small "spraydown" temperature where acid gas removal takes place. Although a number of suppliers offer equipment that could achieve these conditions, very little long-term operating experience and data were available.

For example, a few systems were reported to have operated in Europe and Japan with relatively low outlet temperatures of around 260-275° F (SD/ESP-Zurich), 270-286° F (SD/ESP-Rihuimake), 245-266° F (SD/FF-Leverkusen), and 275-350° F (SD/ESP-Osaka Kantyo). However, in these APC systems, the inlet temperatures were always greater than 350° F, and usually in excess of 400° F (see Table 1). Another APC system operated at low outlet temperatures (Quebec City, 220-275° F), but this system was a pilot plant facility and only operated for a short test period. Only two full scale facilities were operating in the U.S. with limited experience (less than 1 year), Marion using a DS/FF at 280-300° F (outlet) and Commerce using a DS/FF at 280-300° F (outlet).

As will be discussed later, many design and operating parameters impact on successful operation at low spray dryer inlet temperatures. These specific parameters and actual operating conditions were difficult to verify at the time of our selection. In addition, there is reason to believe that the calcium salts problem may be less severe outside the U.S. because of lower chloride content in the waste (in those countries).

PROCUREMENT PROCEDURES

SPECIFICATION DEVELOPMENT

An initial system and equipment specification was prepared by an A&E firm based upon their experience in the power utility field. This specification was modified by Westinghouse and ETS, Inc. engineers. The final product produced a simplified, streamlined bid package with emphasis on those design details thought to be important for this application.

Special attention was made to allow equipment vendors to bid standard product design where possible, yet be specific enough to guarantee an "apples to apples" bid comparison. Although the vendor was being asked for a turnkey bid including warrantees and guarantees based on given site conditions and emission requirements, some design decisions and details were specified by Westinghouse. The type of spray dryer configuration was not defined but a residence time of 15 sec. was required and a pulse jet baghouse with specific design characteristics was specified.

BID SOLICITATION AND REVIEW

Bid responses were solicited from seven vendors by Westinghouse purchasing. A technical review of the seven responses by Westinghouse WESD Environmental Engineering and ETS produced four vendors for final consideration. Three vendors were rejected because of insufficient operating installation experience. The remaining four vendors were asked to make a half day technical presentation to Westinghouse and ETS engineers. The purpose of these technical presentation sessions was to clarify any questions in the bid response, discuss the key design features that affect the operating and maintenance parameters at this specific site, and explore in depth each vendor's past experience on this application. Each vendor was asked to submit a last and final offer. Following a detailed review and spreadsheet comparison, the system proposed by Joy Technologies, Inc. was chosen.

KEY BID EVALUATION CONSIDERATIONS

SPRAY DRYER DESIGN

Of particular concern was the need to operate at the low (350°F) spray dryer inlet temperature. With the calcium chloride dust requiring baghouse operation in the 270°-300°F range to avoid hopper pluggage and bag blinding, the spray dryer only had an operating spraydown temperature of 50°-80°F to scrub out the required SO_2 and HCl gases and evaporate the injected moisture before it enters the baghouse. At the time of this evaluation, no vendor could produce any significant operating experience at the 350°F inlet conditions. Scattered reports of operating problems at low baghouse inlet temperatures were received, but because they were in early start-up phases, the information was sketchy and not generally available. The pros and cons of rotary atomizers vs. dual-fluid nozzles and up-flow vs. down-flow reactors was thoroughly addressed. A rotary atomizer design was chosen as having the best chance of continuously and reliably producing the required slurry atomization to both achieve acid gas efficiencies and minimize liquid carryover and ensure adequate drying.

BAGHOUSE DESIGN

Prior installations had employed both reverse air and pulse jet type baghouses. From a particulate control standpoint, either type system can perform satisfactorily. The pulse jet system was specified because it appears to have the ability to maintain higher incremental acid gas removal. This is due to the ability to retain a thicker filtration cake on the bags following cleaning and in general have a wider range of control over the bag cleaning process through the use of variable pulsing pressure and a combination of on-line and off-line cleaning. The higher available cleaning energy may also be a benefit following those periods when moist or sticky dust is introduced to the baghouse.

SELECTED SYSTEM DESCRIPTION AND FEATURES

The final system layout is shown in Figure 1 and key items listed on Table 2. The APC system consists of a lime preparation system and three independent spray dryer/baghouse trains.

The central lime preparation system is common to all three dry scrubbers. It includes a dry lime storage silo sized for seven days consumption, two complete lime slaking systems each sized to supply the total combined daily reagent requirements of all three spray dryers, one lime slurry storage tank with six hour capacity, and four lime slurry feed systems; one for each spray dryer plus one spare.

The three spray dryers are top inlet, rotary atomizer design sized for a 15 second minimum residence time at design flow. The construction is of 1/4 inch mild steel. The units are arranged with a common penthouse area for maintenance purposes. One spare atomizer assembly is stored in the penthouse.

The three fabric filter systems each consist of six pulse jet modules with full walk-in top plenums. The construction is of minimum 3/16" mild steel. The bags are 6 in. diameters by 16 ft. long on a one piece rigid wire formed cage and the initial bag set will utilize a 16 oz. woven fiberglass fabric with Teflon B finish. The gross A/C ratio is 2.5 to 1 at normal design flow with a net A/C of 3.0 to 1 (one compartment off-line for cleaning). The spray dryer and baghouse use hopper heaters to maintain dust and gas temperatures. All equipment and ductwork is insulated with 4" fiberglass and aluminum box ribbed lagging. Special attention was made to eliminate ambient inleakage and cold spots especially in the hoppers and around access doors.

ASH REMOVAL

Air-operated double-dump discharge valves are located under the spray dryer and baghouse hoppers. The spray dryer has a de-lumper located ahead of the discharge valve. Dust leaving the baghouse discharge valves is conveyed using screw conveyors and emptied into a common drag conveyor for each APC train. Dust from the spray dryer and economizer hoppers is also dumped into the common drag conveyor. The individual drag conveyors (all three APC trains) dump fly ash into a common drag conveyor system which is elevated to either of two fly ash surge hoppers. The hoppers feed twin paddle conditioning screws to moisten the fly ash to a dust-less state. Conditioned fly ash is dumped onto the inclined portion of the bottom ash wet drag conveyor.

INSTRUMENTATION AND CONTROLS

A comprehensive instrument and control system will be employed. Each piece of equipment will have local control and instrumentation to produce automatic system operation based on feedback from continuous emission stack monitors. In addition, all instrumentation and control signals will be fed to a remote Plant Control System for monitoring and over-ride control. All stack CEM data will be continuously available by modem to the PaDER.

CURRENT STATUS OF SIMILAR OPERATING PLANTS

Waste-to-Energy (WTE) facilities brought on-line within the past 18 months have achieved very low emission levels (SO_2, HCl, particulate, heavy metals, and dioxins) during the initial emission compliance test program. It appears that many of these tests were conducted with the APC system operating at conditions to achieve maximum emission reduction, i.e., low outlet temperatures, high lime feed rate, etc. However, based on the recent survey, it appears that many of these same plants have experienced numerous operating problems since initial start-up such as liquid carryover from the spray dryer, bag caking, and plugging of the ash removal system. Additional maintenance and some design changes may be required to solve these operating problems.

In response to a continuing concern regarding trouble-free operation with low inlet tempratures, ETS, Inc. conducted a survey in early 1989 of operating MSW spray dryer absorber/baghouse (SDA/BH) systems. A concerted effort was made to define the actual vs. design operating temperatures, operational problems encountered, and modifications made. Much of the information is closely guarded by the operating facilities and the equipment suppliers; thus the information collected has been treated as confidential by ETS and specific plant sites have not been identified.

The primary finding in this survey was that virtually all SDA/BH sites installed and started up have experienced hopper pluggage and ash removal problems regardless of inlet temperature. Some have also experienced bag caking problems. Most systems having problems have modified the hopper discharge system and some have raised their baghouse operating temperatures.

The issues affecting successful continuous and trouble-free hopper discharge and ash removal are shown below along with a discussion of the survey findings.

1) Effective Drying of Injected Slurry Within the Spray Dryer Vessel
   This requires good control of slurry droplet size and sufficient absorber residence time. Many APC systems are being run over design capacity, thus reducing actual absorber residence time. Lower SDA inlet temperatures (<400°F) makes this objective more difficult especially during periods of operation with worn nozzle inserts for dual-fluid nozzle systems.
2) Flue Gas Temperatures Within the Baghouse
   As the SDA inlet temperature has been lowered, several systems tried to operate at lower baghouse temperatures. The survey indicates a minimum baghouse gas temperature of 270°F is required with 300°F being desirable.
3) Concentration of Calcium Chloride in the Collected Ash
   The higher the concentration of $CaCl_2$ the more difficult the product is to handle. Systems running with higher lime stoichiometry have had less problems. In general, those running with manually controlled lime feed rates, as opposed to automatic SO_2 emission feedback controls, have higher lime usage and less problems. Removing fly ash in cyclones ahead of the spray dryer may also have a negative impact.
4) Adequate Sized Discharge Opening
   Many original hopper discharge systems were designed with 6 to 8 inch openings. The consensus is that a minimum 12" x 12" opening is required with several systems being enlarged to 12" x 24".
5) Eliminating Ambient Inleakage
   Eliminating ambient air inleakages is most critical in order to keep the collected dust from absorbing moisture. Calcium chloride is very hydroscopic at low temperatures and can rapidly absorb moisture from ambient inleakage air turning the dust to a putty or paste. All hopper penetrations and the discharge point must maintain air tight seals. Discharge systems were found ranging from no sealing valve immediately below the hoppers to slide gates plus rotary air locks, or double dump valves at each hopper. Poor sealing with double dump valves due to dust build up on the seals was reported with weekly preventive maintenance required.

6) Maintaining Metal and Dust Temperatures
Hopper heaters and adequate insulation are a must for this application. In addition, the hopper discharge components and ash removal system should also be insulated and possibly heat traced. Large heat sink areas on the hopper walls such as strike plates and poke holes should be heated, insulated, or eliminated. A successful system keeps the ash dry and warm until delivered to its disposal container.

7) Proper Hopper Accessories
Upsets will occur, gas temperatures will drop below minimum limits, SDA nozzles will lose control of droplet size. To overcome these events, properly designed hopper access doors, hopper vibrators, ash level detectors, and poke holes are necessary. However, these items should be designed very carefully to make sure they don't become the cause of additional operating problems instead of tools for good O&M.

8) Properly Sized Ash Conveyor System
Many systems reported undersized ash conveyor systems as a major problem. This problem appears to be caused by:
   a) Some facilities being run over design feed and gas flow rate levels.
   b) Spray dryers running at higher lime feed rates due to high SO_2 and HCl or due to manual feed operating mode.
   c) Higher than expected ash carryover from the combustors.

Both screw and drag conveyor systems are in use and appear viable.

SUMMARY

Resource recovery facility suppliers have designed and successfully operated plants to meet current environmental regulations. More recently, suppliers are designing facilities to meet stringent emission limits while at the same time, attempting to maximize boiler efficiency by operating at reduced economizer outlet temperatures. Westinghouse has designed the York County RRF to operate the boiler with the flue gas temperature leaving the economizer (entering the spray dryer) at 350°F. The APC system will operate at a spraydown temperature of approximately 50 to 80°F and will have to meet very stringent acid gas and particulate emission limits. Westinghouse has specified and selected an APC design that addresses the special O&M conditions resulting from low temperature operation.

In summary, the WTE industry has progressed very rapidly to reduce emissions from these facilities. Facilities have been able to meet very stringent permit limits. Future design attention focused on eliminating reoccurring operating problems will even further improve the industry's environmental performance and overall high plant availability.

TABLE 1

INLET AND OUTLET TEMPERATURES OF VARIOUS AIR POLLUTION CONTROL SYSTEMS INSTALLED ON W-T-E PLANTS

PLANT	APC SYSTEM	INLET TEMP.	OULET TEMP.
Osaka Kankyo	SD/ESP	600	270-340
Quebec City (Pilot Plant)	SD/FF	400-500	220-275
Zurich	SD/ESP	387-480	257-271
Riihimaki	SD/ESP	420-480	260-271
Leverkusen	SD/ESP	392-405	240-260
Kommunekemi	SD/ESP	515-535	265-285
Commerce	SD/FF	500	270-280
Marion	SD/FF	410	280-300

SD - Spray Dryer

ESP - Electrostatic Precipitator

FF - Fabric Filter

TABLE 2

SYSTEM FEATURES AND DESIGN PARAMETERS

SPRAY DRYER

Inlet Designs	Top Roof Gas Disperser
Atomization Type	Rotary Atomizer
Residence Time (sec)	17.95 normal,* 13.9**
Inlet Temp (°F)	350
Inlet Volume (ACFM)	73,404 normal, 95,144 max.
Outlet Temp. (°F)	300-270
Stoichiometry	1.6

BAGHOUSE

Inlet Temp. (°F)	300-270
Outlet Temp. (°F)	290-260
Type	Pulse Jet with Walk-in Plenum
# Modules	6
Bag Size	6" Ø x 16' Lg
Cage	Single Piece, 20 Wires
Fabric	16 oz. Woven Glass/Teflon B
A/C Gross	2.5 normal, 3.2 max.
A/C Net	3.0 normal, 3.8 max.
Pulsing Pressure (psig)	40
Hopper Discharge Flange	12" x 12"
Hopper Heat	4 kw/Hopper

*Normal conditions are defined as those conditions
**Maximum conditions are defined as those conditions

NORMAL	MAX.
50% Excess Air	100% Excess Air
100% Rated Waste Feed	100% Rated Waste Feed
150 ppm - SO_2	300 ppm - SO_2
500 ppm - HCl	1000 ppm - HCl

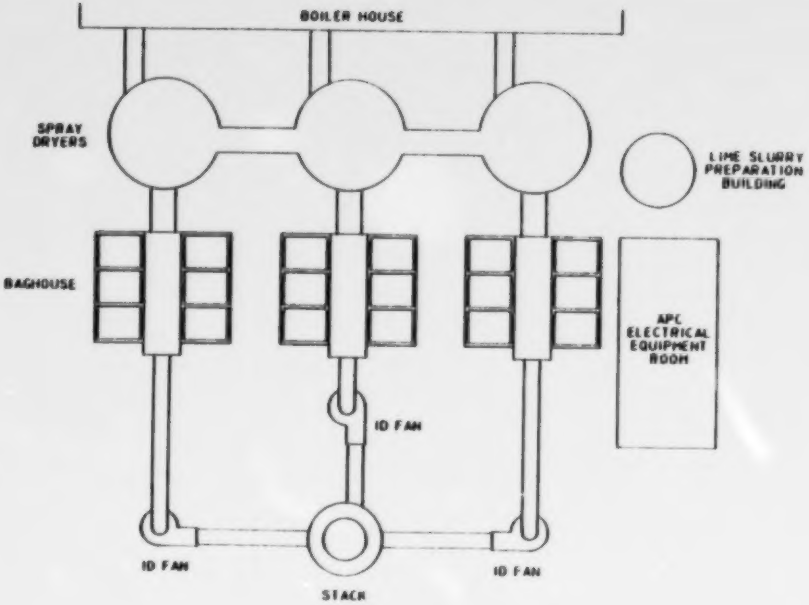
BOILER HOUSE
SPRAY DRYERS
LIME SLURRY PREPARATION BUILDING
BAGHOUSE
APC ELECTRICAL EQUIPMENT ROOM
ID FAN
ID FAN
ID FAN
STACK

FIGURE 1 : YORK COUNTY, PA. RESOURCE RECOVERY FACILITY AIR POLLUTION CONTROL SYSTEM LAYOUT

REVIEW OF DRY INJECTION TECHNOLOGY FOR REDUCING EMISSIONS FROM MUNICIPAL WASTE COMBUSTORS

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ABSTRACT

Dry sorbent injection processes offer an alternative approach for reducing emissions of acid gases and condensible organics from municipal waste combustors. Until recently, however, little performance data has been available from full-scale facilities. This paper reviews several dry sorbent injection processes, their existing applications in North America, selected acid gas and CDD/CDF data, and key factors influencing their economics.

BACKGROUND

On June 26, 1987, the U.S. Environmental Protection Agency issued guidance to State and local environmental agencies calling for application of "dry scrubbers" as Best Available Control Technology (BACT) for municipal waste combustors (MWC's) subject to Prevention of Significant Deterioration (PSD) review under the Clean Air Act (1). At the time this guidance was issued, spray drying was the primary "dry scrubbing" technology used in the U.S. and the only such technology for which control efficiency data were available for MWC's. In Europe and elsewhere, spray dryers are frequently referred to as "wet-dry" scrubbers, reflecting the injection of a wet slurry into the flue gas which dries to produce a dry waste.

This paper reviews several approaches to "dry-dry" scrubbing, their existing applications in North America, selected acid gas and chlorinated dibenzo-dioxins and furans (CDD/CDF) performance data, and key factors influencing their economics.

DESCRIPTION OF DRY-DRY SCRUBBING TECHNOLOGIES

In dry-dry scrubbing, a dry sorbent is injected into the flue gas and a dry waste product is produced. For sake of presentation, dry-dry scrubbing is subdivided into three subgroups: high-temperature furnace injection

(HTFI), moderate-temperature duct injection (MTDI), and low-temperature duct injection (LTDI). This section reviews key design features of these subgroups and identifies existing North American installations.

High-Temperature Furnace Injection

Injection of lime or limestone into the furnace section of an MWC has been used with both conventional and fluidized bed combustor designs. Limestone is less expensive, but requires sufficient furnace temperatures (1,400-2,000°F; 760-1,100°C) and residence times for calcining limestone ($CaCO_3$) to lime (CaO), which in turn reacts with SO_2 and HCl to form $CaSO_3$ and CaCl. In conventional MWC's, limestone can be injected into the furnace using the existing overfire air jets or through separate ports located above the fuel bed. Design requirements for limestone handling are relatively simple.

Furnace injection of limestone has been used with fluidized bed MWC's in Japan and Sweden since the late 1970's (2,3). Excluding limited experimental testing, however, MWC's in Europe and Japan have not used limestone injection with conventional combustors.

In North America (Table 1), furnace injection of limestone is currently used by refractory-lined MWC's in Dayton, Ohio and Davis County, Utah. The waterwall MWC's in Alexandria, Virginia inject lime into the furnace. Limestone injection is also used by fluidized bed MWC's in Tacoma, Washington; LaCrosse, Wisconsin; and Duluth, Minnesota. In each of these cases, the sorbent injection system has been designed and installed by the combustor designer.

Moderate-Temperature Duct Injection

Injection of hydrated lime ($Ca(OH)_2$) at moderate flue gas temperatures (400-600°F, 200-315°C) has been used at MWC's in Japan since the late 1970's. In these facilities, flue gas temperatures are reduced through heat recovery or humidification to protect the particulate control system from high operating temperatures. Sorbent is injected just upstream of the particulate control system. Reaction of the sorbent and acid gases is rapid, allowing injection of sorbent as little as one to two seconds upstream of the particulate matter control system. In excess of 100 MWC's in Japan are reported to use MTDI of hydrated lime, primarily to meet HCl emission limits (2,11).

North American installations (Table 2) injecting hydrated lime at moderate temperatures include Claremont and Concord, New Hampshire, and Dutchess County, New York. These three facilities use heat recovery as the primary means for reducing flue gas temperature. An automatically controlled flow of ambient air is used at the two New Hampshire facilities to protect the fabric filter from high temperature excursions. Sorbent is injected through a venturi located just prior to the fabric filter. The MWC facility in Sitka, Alaska injects sodium bicarbonate ($NaHCO_3$) at 450-500°F (240-270°C) between the unit's waste heat boiler and ESP.

An alternate duct injection process involves injection of sodium bicarbonate into the convection section of the heat recovery system at flue gas temperatures of 520-1,560°F (290-850°C). At these temperatures, sodium bicarbonate decomposes to sodium carbonate (Na_2CO_3) which reacts with HCl and SO_2 in the flue gas. In addition to reducing acid gas emissions, the process vendor claims that sufficient chloride is removed from the flue gas to limit downstream formation of chlorine-dependent CDD/CDF (17).

Low-Temperature Duct Injection

Injection of hydrated lime at lower flue gas temperatures (<300°F; 150°C) is a relatively new process for reducing emissions from MWC's. By reducing flue gas temperatures to these lower temperatures, it is possible to increase sorbent reactivity as well as increase removal of condensible metals and organics. Flue gas temperatures can be reduced through indirect heat exchange (to air or water) or humidification. Sorbent is injected through a venturi or into a reactor vessel just upstream of the particulate control system. Operating temperatures in the particulate matter control system are similar to those found in spray dryer scrubbing systems. However, because there is no need to evaporate moisture from the sorbent, flue gas residence time prior to the particulate matter control system can be reduced to 1-2 seconds versus 10-15 seconds in a spray dryer system. Commercial application of this technology began in Europe in the early 1980's (18). North American vendors of LTDI systems include Flakt, Interel, Procedair, and Engineering Technology Group.

Low-temperature duct injection systems in North America (Table 3) are currently operating or under construction in Springfield, Massachusetts; St. Croix, Wisconsin; Muscoda, Wisconsin; Vancouver, British Columbia; London, Ontario; and Quebec City, Ontario. All of these systems use a fabric filter to capture particulate matter and condensibles.

COMPARATIVE ACID GAS AND CDD/CDF PERFORMANCE DATA

This section reviews acid gas and CDD/CDF emissions data collected during compliance tests at four MWC's with dry sorbent injection systems: Alexandria, Virginia (equipped with HTFI/ESP); Claremont, New Hampshire (MTDI/FF); St. Croix, Wisconsin (LTDI/FF); and Springfield, Massachusetts (LTDI/FF). These data are compared to emissions data from spray dryer-equipped MWC's in Commerce, California (SD/FF); Marion County, Oregon (SD/FF); and Millbury, Massachusetts (SD/ESP). Selected data collected during parametric testing at Quebec City, Ontario (LTDI/FF and SD/FF) and Dayton, Ohio (HTFI/ESP and MTDI/ESP) are also presented.

Key process, acid gas, and CDD/CDF data collected during the the above tests are shown in Table 4. Although the design and operation of these sorbent injection systems are different, certain similarities and differences are worth noting.

The effect of sorbent injection method and flue gas temperature on SO_2 emissions is shown in Figure 1. For each of the injection methods, as well as overall, emissions increase as flue gas temperature increases. No clear distinction exists between SO_2 emission levels as a function of sorbent injection location (duct or furnace), other than as is reflected by flue gas temperature. The lowest SO_2 emissions result from LTDI while the highest are for MTDI. These data suggest that the flue gas temperature in the PM control device has a major impact on SO_2 emissions. Sorbent feed rates (which are not plotted in Figure 1) also influence SO_2 emission rates.

In Figure 2, the effect of flue gas temperature on HCl emissions for the different injection methods is shown. For all three injection methods, emissions generally increase as temperature increases. For a given temperature, however, HCl emissions are highest with furnace injection.

The relationship between SO_2 and HCl emissions as a function of injection method is shown in Figure 3. For LTDI and MTDI, HCl emissions are lower than or similar to SO_2 emissions. Because uncontrolled acid gas levels are generally higher for HCl than for SO_2 (typical U.S. acid gas levels are 300-500 ppm for HCl versus 100-200 ppm for SO_2), this suggests that removal efficiencies associated with duct injection are significantly greater for HCl than for SO_2. For HTFI, HCl emissions are consistently higher than SO_2 emissions, suggesting that removal efficiencies for SO_2 and HCl are roughly equal.

To assess the impact of dry injection system configuration on CDD/CDF removal, three key factors were examined: (1) the method of dry scrubbing, (2) the type of particulate control device, and (3) the flue gas temperature. In Figure 4, 2378-TCDD toxic equivalency factors (TEF's; U.S. EPA method) are plotted as a function of flue gas temperature and control technology. Based on 2378-TCDD TEF values, CDD/CDF emissions are consistently higher with ESP's than with fabric filters for both wet-dry and dry-dry scrubbing. At 250°F (120°C), TEF levels are 90 percent lower for systems equipped with a fabric filter than for similar systems with an ESP. Increasing flue gas temperature increases CDD/CDF emissions. Although TEF values appear somewhat lower for spray dryers than for dry injection systems operating at similar temperatures, these differences are relatively small when compared to the impact of the different PM control devices.

ECONOMIC CONSIDERATIONS

The economics of a sorbent injection emission control system will depend upon the size of the facility, method of flue gas cooling, regulatory requirements, and other factors. Due to its fairly simple hardware requirements and lower sorbent costs, injection of limestone or lime into the furnace will generally be the least expensive alternative from both a capital and annualized cost bases.

The relative cost of duct injection of dry sorbent versus spray drying depends on the size of the facility and method of flue gas cooling. Although comparative cost data are limited, duct injection systems are generally lower in capital cost (due to the elimination of slurry preparation and handling facilities), but higher in operating costs (due to the higher cost of hydrated lime) than spray dryer systems. In general, duct injection systems become less economically attractive as the size of the combustor increases. However, for MWC's that can reduce flue gas temperature while recovering of useable heat (e.g., with a combustion air preheater or low-temperature steam economizer) rather than through evaporative cooling as with a spray dryer, duct injection systems may retain their economic attractiveness with even fairly large combustors.

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TABLE 1. FACILITIES WITH HIGH-TEMPERATURE FURNACE INJECTION

Facility/Location	Number of Units/TPD	Start-up Date	Vendor	Sorbent	PM Control[a]	Flue Gas Temperature[b] (°F)	Reference
Montgomery County North/ Dayton, Ohio	3 @ 300	1970, 1988[c]	Self	Limestone	ESP	550	4
Montgomery County South/ Dayton, Ohio	3 @ 300	1970, 1988[c]	Self	Limestone	ESP	550	4,5
Davis County, Utah	2 @ 200	1987	Katy-Seghers	Limestone	ESP	420	6
Alexandria, Virginia	3 @ 325	1987	Ogden-Martin	Lime	ESP	360	7
Tacoma, Washington	2 @ 330	1988	Energy Products of Idaho	Limestone	FF	350	8
LaCrosse County, Wisconsin	2 @ 200	1987	Energy Products of Idaho	Limestone	EGB	325	9
Duluth, Minnesota	2 @ 200	1986	Copeland	Limestone	Wet Scrubber	400	10

[a] ESP – Electrostatic precipitator; FF – fabric filter, EGB – electrified gravel bed.

[b] Flue gas temperature at the PM control device inlet.

[c] Two units started in 1970; the third started in 1988. Limestone injection retrofit onto Units 1 and 2 in 1988.

TABLE 2. FACILITIES WITH MODERATE-TEMPERATURE DUCT INJECTION

Facility/Location	Number of Units/TPD	Start-up Date	Vendor	Sorbent	PM Control[a]	Flue Gas Temperature[b] (°F)	Reference
Claremont, New Hampshire	2 @ 100	1987	Wheelabrator	Hydrated Lime	FF	450	12,13
Concord, New Hampshire	1 @ 500	1989	Wheelabrator	Hydrated Lime	FF	NA[c]	14
Dutchess County, New York	2 @ 253	1988	Westinghouse	Hydrated Lime	FF	375	15
Sitka, Alaska	2 @ 25	1985	Sigoure/Freres S.A.	Sodium Carbonate	ESP	450	16

[a] FF - fabric filter, ESP - electrostatic precipitator.

[b] Flue gas temperature at the PM control device inlet.

[c] NA - not available.

TABLE 3. FACILITIES WITH LOW-TEMPERATURE DUCT INJECTION

Facility/Location	Number of Units/TPD	Start-up Date	Vendor	Sorbent	PM Control[a]	Flue Gas Temperature[b] (°F)	Reference
Springfield, Massachusetts	3 @ 120	1988	Flakt	Hydrated Lime	FF	280	19
St. Croix, Wisconsin	3 @ 40	1988	Interel	Hydrated Lime	FF	225	20
Muscoda, Wisconsin	2 @ 50	1989	Interel	Hydrated Lime	FF	NA[c]	21
London, Ontario	3 @ 100	1987	Procedair	Hydrated Lime	FF	300	22
Vancouver, British Columbia	3 @ 350	1988	Flakt	Hydrated Lime	FF	300	19
Quebec City, Ontario	4 @ 250	1989	Procedair	Hydrated Lime	FF	300	22

[a] FF = fabric filter.

[b] Flue gas temperature at the PM control device inlet.

[c] Not available.

TABLE 4. SELECTED DATA FROM MASS BURN COMBUSTORS WITH DRY SORBENT INJECTION OR SPRAY DRYING

Facility/Location	Injection Method[a]	Sorbent	PM Control[b]	Flue Gas Temperature[c]	SO_2[d] (ppm, dry @ 7% O_2)	HCl[d] (ppm, dry @ 7% O_2)	2378-TCDD Toxic Equivalent[d,e] (ng/dscm @ 7% O_2)
Dayton, OH	HTFI	Limestone	ESP	300	21(82)	35(88)	NA
Alexandria, VA	HTFI	Lime	ESP	360	37	170	0.74
Dayton, OH	HTFI	Limestone	ESP	400	57(49)	120(58)	NA[f]
Dayton, OH	MTDI	Hydrated Lime	ESP	400	55(52)	34(88)	NA
Quebec City, ON	MTDI	Hydrated Lime	FF	400	95(23)	110(76)	0.11(99)[g]
Claremont, NH	MTDI	Hydrated Lime	FF	450	110	55 (92)	0.33
St. Croix, WI	LTDI	Hydrated Lime	FF	225	33	0.75	0.044
Quebec City, ON	LTDI	Hydrated Lime	FF	230	4.8(96)	7.4(98)	0.037(99)[g]
Springfield, MA	LTDI	Hydrated Lime	FF	280	22(83)[h]	31(94)[h]	0.14[h]
Dayton, OH	LTDI	Hydrated Lime	ESP	300	39(67)	8.9(97)	NA
Millbury, MA	SD	Lime Slurry	ESP	250	58(76)	15(98)	0.49(64)
Commerce, CA	SD	Lime Slurry	FF	270	1.2(99)	8.8(99)	0.0032(88)
Marion County, OR	SD	Lime Slurry	FF	280	120(71)	35(95)	0.052(96)
Quebec City, ON	SD	Lime Slurry	FF	280	42(63)	38(92)	0.0096(99)

Footnotes continued -

[a] HTFI - high-temperature furnace injection; MTDI - moderate-temperature duct injection; LTDI - low-temperature duct injection; SD - spray dryer.

[b] ESP - electrostatic precipitator; FF - fabric filter.

[c] Flue gas temperature at the PM control device inlet.

[d] SO_2, HCl, and 2378-TCDD toxic equivalent emissions reported. Values in parentheses indicate calculated removal efficiencies based on actual inlet values. Where an inlet concentration was unavailable, no removal efficiency was calculated.

[e] 2378-TCDD toxic equivalent based on U.S. EPA method.

[f] NA - not available.

[g] 2378-TCDD toxic equivalent approximated from total tetra- through octa-chlorinated dioxins and furans concentration and a correlation between the two measures (23).

[h] Emissions normalized to 12 percent CO_2.

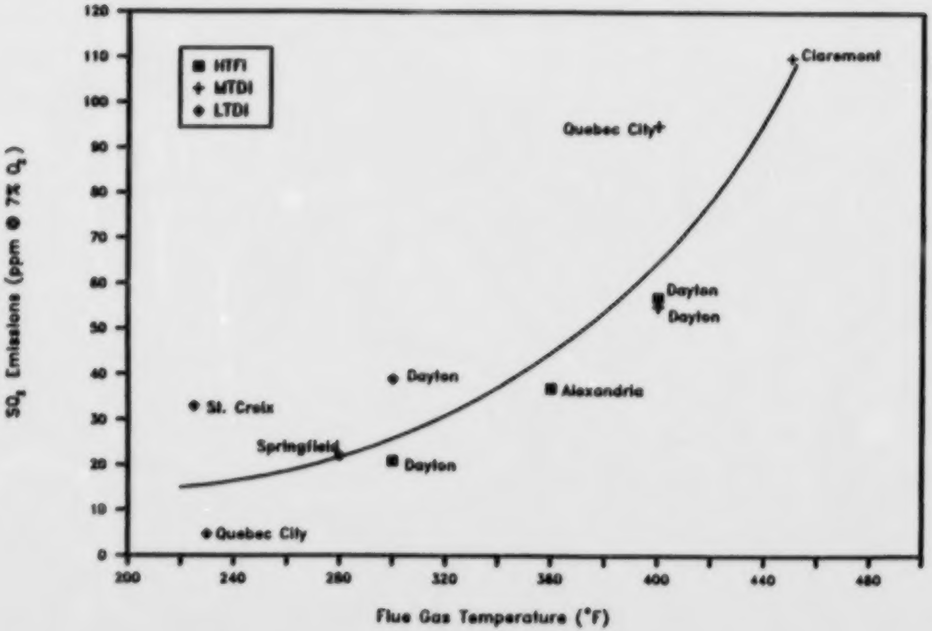

Figure 1. Effect of Sorbent Injection System and Flue Gas Temperature on SO_2 Emissions

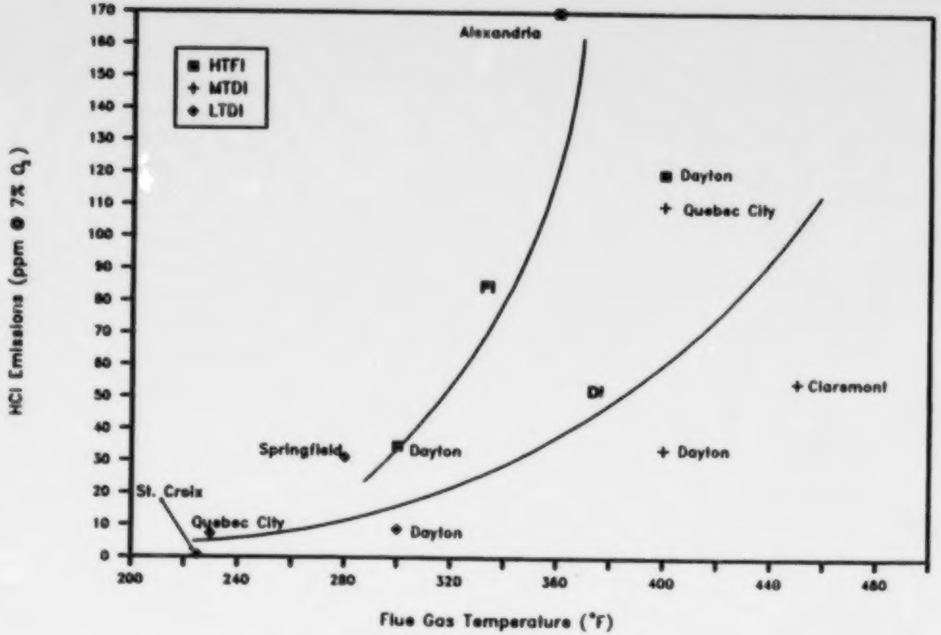

Figure 2. Effect of Sorbent Injection System and Flue Gas Temperature on HCl Emissions

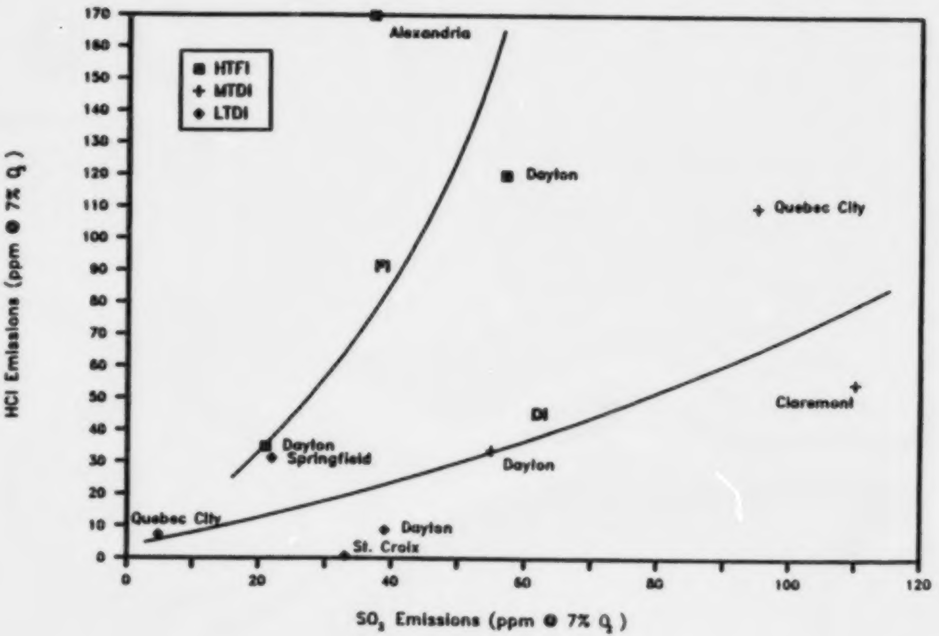

Figure 3. Relative SO_2 and HCl Emissions for Dry Injection Systems

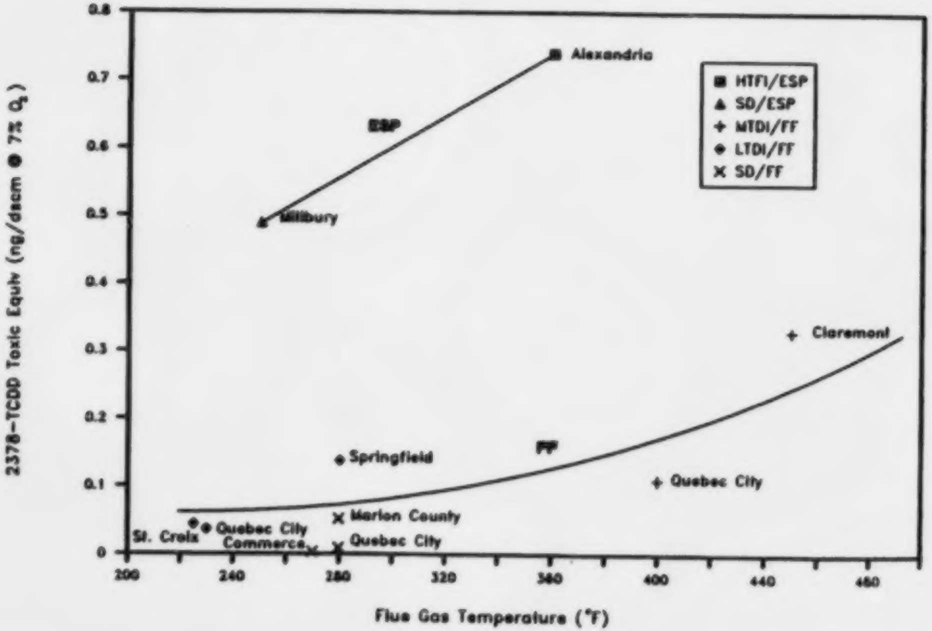

Figure 4. Effect of Flue Gas Temperature and Particulate Control Device on CDD/CDF Emissions

RECOMMENDED QA/QC ACTIVITIES FOR MUNICIPAL WASTE COMBUSTOR TESTING

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ABSTRACT

Sampling and analysis of air emissions and waste streams from municipal waste combustors (MWCs) typically include protocols for trace metals, trace organics, nitrogen oxides (NO_x), carbon monoxide (CO), sulfur dioxide (SO_2), and hydrogen chloride (HCl). Several of the sampling and analysis methodologies employed for MWC testing are currently undergoing development and validation on federal level to meet the testing requirements for state and local agencies. To produce valid data, each sampling and analytical protocol should include comprehensive QA/QC procedures. The recommended QA/QC procedures are discussed in this paper for the following sampling methodologies: flue gas trace metals, flue gas trace organics, NO_x, CO, refuse-derived fuels (RDF), ash discharge streams, SO_2, and HCl. The recommended QA/QC procedures are discussed for the following analytical protocols: trace metals, trace organics, and instrumental analyses (NO_x, CO, SO_2, and HCl).

This paper has been reviewed in accordance with the U. S. Environmental Protection Agency's peer and administrative review policies and approved for presentation and publication.

INTRODUCTION

Municipal waste combustion is increasingly becoming the preferred technique for disposal of garbage in both the United States and Canada. In response to concern for the health and environmental effects of the emissions and waste steams from municipal waste combustion facilities, both the U. S. Environmental Protection Agency (EPA) and Environment Canada are currently sponsoring major studies to assess these byproducts from municipal waste combustors (MWCs). The primary focus of the U. S. EPA study is to supply supporting data for development of regulations for MWCs under Section 111 of the Clean Air Act, while the main objective of Environment Canada's National Incinerator Testing and Evaluation Program (NITEP) is to define optimum design and operating conditions to minimize the emissions of concern. This mutual interest has prompted an exchange of data between the two agencies and, most recently, a cooperative test program designed to fulfill the objectives of both agencies.

Because of the importance of the results, the expense of the testing, and the quantity of data being collected, quality assurance and quality control (QA/QC) have been stressed in all areas of testing, thereby ensuring that all data are of the highest possible documented quality. Because of the need to maintain high data quality for these MWC test programs, external QA/QC programs have been instituted to assess the data quality using personnel totally independent of those involved in making measurements.

To provide an understanding of the complexity of these MWC emission test programs and the need for a comprehensive QA/QC program, an overview of the MWC studies along with the basis for QA/QC is presented. This is followed by a discussion of the design and implementation of a comprehensive external QA/QC program for MWC testing, including pertinent examples from the external QA/QC program conducted recently for the joint Environment Canada/U. S.-EPA sponsored municipal waste combustion and emission evaluation program at the Mid-Connecticut Resource Recovery Facility in Hartford, Connecticut and other EPA-sponsored test programs.

QA/QC DEFINITIONS

Quality assurance and quality control are defined somewhat differently for different objectives. For the purpose of air pollution studies, they may be differentiated by stating that quality control is "the system of activities to provide a high quality product," whereas quality assurance is "the system of activities to provide assurance that the quality control system is performing adequately."[1]

Quality control may be considered as "internal quality control:" routine checks included in normal internal procedures. Examples are periodic calibrations, duplicate checks, split samples, and spiked samples. Quality assurance is then considered as "external quality control" and includes those activities that are performed on an occasional basis, usually by personnel outside the normal routine operations. Examples of these activities are on-site system surveys or technical system audits, independent performance audits, interlaboratory comparisons, and periodic evaluation of internal quality control data. Thus, the objective of QA/QC for emission testing programs is to improve or assure the quality of the measured data. Since emission measurements are made by numerous agencies and private organizations, QA/QC activities are also concerned with establishing and assessing the comparability of data for use in data bases. Under a QA/QC program, data quality is typically quantified (where possible/practical) and reported using the following five key QA specifications:

- precision,
- accuracy,
- representativeness,
- completeness, and
- comparability.

Precision is defined as the mutual agreement among individual measurements of the same property. Precision estimates are often obtained by collocating sampling systems, replicate analysis, replicate sampling, etc.

Accuracy is defined as the degree of agreement between a measured value and an accepted reference or true value. Analytical accuracy is often determined by analyzing standard reference materials (SRMs). For example, accuracy of a flow measurement system may be determined using a critical orifice. These determinations for QA purposes are called performance evaluation audits.

Representativeness expresses the degree to which data accurately and precisely represent a characteristic of a population, a process condition, or an environmental condition.

Completeness is a measure of the amount of valid data obtained compared to the amount of data expected under normal conditions.

Comparability expresses the confidence with which one data set can be compared with another.

SIGNIFICANCE OF QA/QC PROGRAMS FOR MWC TESTING

In July 1987, the U. S. EPA published a notice of proposed rulemaking in the Federal Register which describes emission standards development for MWCs under Section 111 of the Clean Air Act (Standards of Performance for New Sources): for new MWCs under Section 111(b) and for existing MWCs under Section 111(d). The notice culminated 18 months of work resulting in the development of the technical and health-related documents comprising EPA's Report to Congress on MWCs,[2] which was a joint effort involving the Offices of Air Quality and Planning and Standards (OQAPS), Solid Waste (OSW), and Research and

Development (ORD). OAQPS is presently responsible for reviewing existing data and gathering new data, particularly regarding emerging pollution control technologies, to create an MWC emission data base. As a result, OAQPS has sponsored a number of MWC emission test programs in support of the current standards development work.

Municipal waste combustion, because of its heterogeneous and variable fuel feed, emits a variety of pollutants. The emissions being studied in the overall assessment of MWCs include the criteria pollutants -- particulate matter (PM), sulfur dioxides (SO_2), nitrogen oxides (NO_x), carbon monoxide (CO), total hydrocarbons (THC), and lead (Pb); other acid gases, such as hydrogen chloride (HCl); trace chlorinated organics, including polychlorinated dibenzo-p-dioxins (CDD), polychlorinated dibenzofurans (CDF), and CDD/CDF precursors; and specific metals including arsenic (As), cadmium (Cd), total chromium (Cr), mercury (Hg), and nickel (Ni).

To date, OAQPS has sponsored or co-sponsored a total of ten test programs at:

- One mass-burn MWC with ESP control;
- Two mass-burn MWCs with spray dryer/fabric filter control;
- One mass-burn MWC with spray dryer/ESP control;
- One refuse-derived fuel MWC with spray dryer/fabric filter control; and
- One mass-burn MWC with in-furnace limestone injection and ESP control. (Lime injection into the flue gas duct upstream of the ESP was also studied in this program.)

All of these programs have been quite complex, involving various combinations of air emission sampling for multiple pollutants, simultaneous sampling at multiple locations to allow control efficiency determinations, use of unconventional and state-of-the-art manual and instrumental test methods, and process stream sampling. In addition to the different facilities, a number of different American and Canadian contractors have been utilized to perform the emission testing. Thus, it was recognized that rigorous QA/QC efforts are essential to ensure successful test programs in terms of data quality (achieved accuracy, precision, completeness and representativeness) and data comparability. For the most complex test programs, external QA/QC programs have been established. In these cases involving collection of hundreds of individual samples and continuous hours of emission data monitoring over a one-to-six week time span, it was considered prudent to delegate the QA activities to an independent third party.

RECOMMENDED MWC TEST PROGRAM QA/QC ACTIVITIES

QA/QC activities to establish, qualify, and/or quantify the data quality have been applied at all phases of each MWC test program: the planning and pre-test phase, the sampling phase, the analytical phase, and the data reduction/reporting phase. The following discussions summarize the QA/QC activities applied to each phase. It is strongly recommended that parallel

QA/QC programs be applied to future MWC emission testing efforts to ensure the highest data quality and data comparability.

PLANNING AND PRE-TEST QA/QC ACTIVITIES

QA/QC activities carried out prior to the commencement of the sampling include test plan development, QA plan development, equipment calibration, sampling equipment and sample container cleaning, reagent quality checks, and standardization of sample integrity and documentation procedures.

Development of the test plan and the QA plan promotes the early definition of test and QA objectives, respectively. The test plan includes detailed discussions of all sampling and analytical procedures, accessibility and representativeness of sampling locations, a process description, data reduction and reporting, internal QA/QC, safety considerations, and, of course, test objectives. In almost every case, it is necessary to visit the test site prior to preparing the test plan. This visit allows inspection of each sampling location, confirmation of process parameters, and determination of needed site modifications. An outline for a typical test plan is shown in Figure 1.

A QA plan deals specifically with the QA/QC aspects of the test program. A separate QA plan should be prepared for all but the simplest MWC test programs. It is advantageous for preparation of the QA plan that the test plan has been finalized. QA plans are typically prepared following the guidelines issued by EPA's Office of Research and Development,[3,4] which ensure that all elements of a comprehensive QA program are addressed. Specifically, the 14 elements of a QA plan are:

- Project Description
- Project Organization and Responsibility
- QA Objectives
- Sampling Procedures (QA/QC Aspects)
- Sample Custody
- Calibration Procedures and Frequency
- Analytical Procedures (QA/QC Aspects)
- Data Reduction, Validation, and Reporting
- Internal QC Checks
- Performance and System Audits
- Preventive Maintenance
- Assessment of Data Precision, Accuracy, and Completeness
- Corrective Action
- Reports to Management

The preparation of a QA plan compels consideration of all these elements and how they will interact to accomplish the QA objectives. The QA objectives for a test program are often expressed in terms of specific numerical goals for precision, accuracy, and completeness of the data, which are based on experience in application of the particular measurement systems. The typical objectives for the EPA-sponsored test programs are summarized in Table 1. The

QA plan also typically defines the acceptance criteria and control limits for the various sampling and analytical procedures (see Table 2). It is also very beneficial, as shown in Table 2, to delineate the appropriate corrective actions for these criteria.

All field equipment must be properly calibrated, and these calibrations documented prior to the sampling program. This includes manual sampling equipment (pitot tubes, nozzles, temperature devices, barometers, dry gas meters, flowmeters), balances, and cylinder gases.

To ensure that samples are not contaminated, sampling equipment components and sample containers must be cleaned in a manner compatible with the sample materials with which they will come in contact, and checks of reagent quality must be performed.

Particularly important for complex test programs conducted at MWC's are: (1) development of a logical sample numbering scheme and a system for sample handling and processing and (2) assembly of a set of field data sheets which provide for documentation of all significant data.

ON-SITE QA/QC ACTIVITIES

As would be expected, the majority of the QA/QC activities are accomplished on-site during the testing phase. These include:

- o calibration of the continuous emission monitors;
- o technical systems audits and performance evaluation audits of manual emission and process stream sampling, instrumental measurements, and process data correction;
- o internal QC, which is performed by the test personnel, but is monitored externally using the technical systems audits; and
- o periodic data review.

These activities are described below in relation to manual emission sampling, continuous instrumental sampling, process stream sampling, and process data collection.

On-Site QA for Manual Emission Sampling

Test personnel conducting the sampling perform the internal QC checks associated with manual sampling. These include:

- o confirming proper assembly and sealing of sampling trains;
- o pretest, port-change, and post-test leak checks;

- stability of sampling train component temperatures;
- checking cyclonic flow at sampling location; and
- collection of field, recovery, and proof rinse blanks.

Technical systems audits of the manual sampling systems are typically conducted by personnel not involved in making the measurements. Technical systems audits are a qualitative evaluation of all components of a measurement system to determine proper selection and use. Components of a measurement system include on-site facilities, equipment, recordkeeping, data validation, operation, maintenance, calibration procedures, reporting requirements, and internal QC procedures. For the EPA-sponsored MWC test programs, each type of manual sampling system (for example, Modified Method 5 for CDD/CDF) has been subjected to a full technical systems audit at least once during the test program. Internal QC checks, such as leak checks and collection of blank samples, are typically observed by the external QA personnel each time they occur. Specific data sheets for the various sampling systems (see Figure 2 for an example) have been developed for use in performing thorough on-site technical systems audits.

Performance evaluation audits, which quantitatively evaluate the accuracy of certain aspects of a measurement system, are performed on-site for manual emission sampling to verify the accuracy of the temperature devices and the dry gas meters in the meter boxes. Audit materials and acceptance criteria are included in Table 3.

On-site QA for manual sampling also includes review of the data collected for each sampling run. This review for completeness, consistency, and reasonableness is generally performed at the end of each test day.

On-Site QA for Instrumental Measurements

Recent MWC programs have relied heavily on the use of instrumental emission measurements, where possible, to provide real time information on concentrations and emission trends which can also be compared with process parameter trends. Pollutants measured using instrumental techniques have included NO_x, SO_2, CO, THC, and HCl. O_2 and CO_2 levels have also been measured instrumentally. At the present time, instrumental test methods have been promulgated for NO_x, SO_2, CO, O_2/CO_2, and THC; these are EPA Methods 7E, 6C, 10, 3A, and 25, respectively. Because of the long run times used for the MWC test programs, the performance specifications required by these methods have been relaxed for the EPA sponsored test programs (see Table 2) in a compromise between the instrumental methods and continuous emission monitoring.

When emission measurements are made instrumentally, the instruments must be calibrated on-site. This calibration should involve at a minimum a two-point calibration (zero and span) prior to each test run, coupled with a second two-point calibration at the end of each run to assess and quantify instrument drift. For the EPA-sponsored test programs, when the drift was over 5 percent the monitoring data was drift corrected. Drift results greater than 20 percent were considered unacceptable, and the data either were rejected or drift corrected and retained conditionally.

If suitable calibration gases are available, a multipoint (three- or four-point) linearity check is conducted prior to the start of testing and once every two to three days afterward during the test program.

QC checks are typically conducted at least once a day for each instrument. Control standards (certified gases, ± 2%) are introduced upstream of the sample conditioning systems, and the acceptance criterion is that the system response must be within 10 percent of the certified gas value or the overall (running) mean for previous QC checks.

Unless prior knowledge of the instrumental sampling locations shows it unnecessary, stratification checks should be performed. Acceptable stratification is when all points of a traverse yield responses within 10 percent of their average.

Interference checks have also been typically performed for the CO_2, O_2, and SO_2 analyzers. Interferences for CO_2 and O_2 are checked by comparing the instrumental data with that from simultaneous EPA Method 3 testing; interference for SO_2 is checked against EPA Method 6.

The on-site technical systems audits for the instrumental measurements involve verification of the calibrations, drift checks, QC checks, linearity checks, stratification checks, and interference checks, along with careful observation of the procedures in routine operation of the instruments and data acquisition systems.

On-site performance evaluation audits are also performed on the instrumental measurement systems. Typically, each instrumental system is subjected to a performance audit prior to, midway through, and at the end of the test program, where the response for a Protocol 1[5] audit gas has to be within 10 percent of the audit value. System bias checks of each monitoring system are also performed as performance evaluation audits at the same frequency. For these audits, the gases are introduced into the probe rather than directly into the analyzer to assess the combined sampling system and analyzer bias. The acceptance criterion for the percent error for both the analyzer accuracy audits and the system bias checks is ± 10 percent. Measurements are evaluated at the analyzer output as well as the data acquisition recording station output.

There are currently no performance specifications for HCl CEMSs and no instrumental HCl method. In addition, at the time of the EPA-sponsored testing, a range of certified HCl gas cylinders was not available for use in QA/QC activities on HCl monitors. Therefore, routine multipoint calibrations and performance evaluation audits could not be conducted for these instruments. Relative accuracy audit tests were used instead of cylinder gas audits to establish the accuracy of these instruments. The results of wet chemical (impinger train) sampling for HCl were used as the "true" or reference values. The monitor values had to agree within 25 percent of the reference values.

At the end of each day of testing, all the instrumental data are reviewed for completeness, reasonableness, and consistency.

On-Site QA for Process Stream Sampling

On-site technical systems audits for the process stream sampling involve observation of all the steps in the collection and handling of each type of process sample. This includes determining if the sampling equipment is suitable, if the sampling locations are representative, and that cone and quartering procedures will produce homogeneous composites.

On-Site QA for Process Data Collection

On-site technical systems audits for process data collection have usually involved two phases. The first is simple observation of the data collection procedures to confirm that the specified data are being collected from the proper output. Secondly, the primary set of operational data averages is compared to similarly generated averages obtained independently over the same time period. These should agree within 10 percent. For one test program, certain strip chart results were digitized. An audit of the digitization involved comparison of the averages generated from the digitization to averages generated from manual collection of the same data over the same period. This audit revealed several problems with the digitization procedures that were resolved on-site.

For another test program in which a central data acquisition system (DAS) was used to collect and summarize the process data, a technical systems audit of the DAS involved comparison of the DAS output to averages generated from manual collection of parallel data during the same time period.

ANALYTICAL QA/QC ACTIVITIES

Key elements in QA/QC of the analytical phase of a test program are use, wherever possible, of suitable, accepted laboratory protocols with prescribed QC procedures and thorough prior review of new or unconventional protocols. Analytical QC typically focuses on the use of control standards to provide an internal measure of precision and accuracy. Also, specific acceptance criteria are defined prior to the test program for the various analytical operations including calibrations, control standard analyses, blanks, etc.

For example, for analysis of flue gas and ash metals samples, laboratory QC generally involves analysis of interference check samples, matrix spike samples, and duplicate samples.

The analysis for trace organic compounds such as CDD/CDF using gas chromatography/mass spectroscopy is a highly specialized procedure requiring a number of QC checks and analytical criteria to ensure quality data:

- Labelled Internal Standards - Added to all samples prior to extraction, clean-up, and analysis; recovery results used to adjust results of native species reported and to assess efficacy of these procedures (see Table 2 for acceptance ranges).

- Labelled Surrogate Standards - Added to XAD-2® resin cartridges before sampling; recoveries used to evaluate the sampling and extraction efficiency (see Table 2 for acceptance ranges).

- Labelled Recovery Standards - Added to each sample extract prior to injection into the GC/MS to directly assess the instrumental analysis; results are used in the calculation of the internal standard recoveries.

- Field Blanks - Exposed to field conditions and analyzed to assess possible field contamination.

- Method Blanks - Prepared in laboratory and analyzed to assess possible laboratory contamination.

- Reagent Blanks - Prepared and analyzed to determine background contamination of the reagents/solvents used in sampling and analysis.

- Proof Rinse Blanks - Prepared by rinsing sampling equipment and sample containers with sampling and/or rinse reagents and analyzed to assess contamination of these items.

- Analytical Criteria - Used to confirm target trace organic compounds:
  - Retention time,
  - Chlorine isotope ratio of molecular ions, and
  - Signal-to-noise ratio.

The major QC procedures for the Volatile Organic Sampling Train (VOST) analysis include GC/MS tuning; calibration system performance checks; analysis of QC check samples; and analysis of a reagent blank, matrix spike, and a matrix spike duplicate.

Performance evaluation audits are the principal QA activities associated with the laboratory phase and are used to assess analytical accuracy. In the few cases where samples can be split (e.g., metals, HCl, ash), they are used to assess the between-laboratory precision.

For metals analysis, accuracy is assessed for the front- and back-half samples, respectively, using spiked filters and spiked liquid samples. For mercury and Cr^{+6} analyses which do not include a filter, only spiked liquid samples are used. The target limits for accuracy for all these audit samples are ± 10 percent.

The audit sample used to assess the ion chromatographic analysis for manual HCl sampling is also a liquid with a target limit for accuracy of ± 10 percent.

For the CDD/CDF analysis, spiked XAD-2® audit samples, which recently became available from the Quality Assurance Division (QAD) of EPA's Atmospheric Research and Exposure Assessment Laboratory, are used to assess accuracy. The target limits for this analysis are the results for all isomer/isomer groups within 50 percent of the 90 percent confidence interval from previous analyses.

The VOST analysis is audited using Group 1 cylinder gases at ppb levels, also available from QAD. These gases are sampled using the VOST train in the field or the laboratory and the analysis performed as per the field samples. The target limits are ± 50 percent of the cylinder concentration.

A new audit material is now available from QAD to check the particulate recovery from particulate/trace metal sampling trains. This device contains a known amount of NBS fly ash (No. 1633a) which is introduced into the nozzle of the train. The sample is then recovered and analyzed gravimetrically. The results should be within ± 10 percent of the audit value. The NBS fly ash may also be analyzed for metals as an additional assessment of the accuracy of the metals analysis.

DATA REDUCTION/REPORTING QA/QC ACTIVITIES

All data are validated prior to final presentation. Data validation is a systematic procedure of reviewing data against a set of established criteria to provide a level of assurance of its validity prior to its intended use. Validation criteria include: (1) process conditions during sampling or testing, (2) adherence to control limits shown in Table 2, (3) acceptable performance evaluation and technical systems audit results, and (4) consistency and reasonableness based on a knowledge of site characteristics and sampling locations.

As part of the data validation process, audits for data quality are also conducted. These audits include the detailed review of: (1) the recording and transfer of raw data; (2) data calculations; (3) the documentation of procedures; and (4) the selection and use of appropriate data quality indicators. Some of the data quality indicators used are:

- Comparison of the F_o for all instrumental CO_2 and O_2 data and Method 3 data;
- Comparison of process monitoring O_2 and instrumental O_2 data.
- Comparison of the measured volumetric flow rate from different sampling locations;

- Comparison of the control system's instrumentation temperature readings and the manual methods readings;
- Comparison of the measured volumetric flow rate with the calculated flow rate using the tons of refuse burned, steam load, and the excess air;
- Comparison of the relative concentrations of the emissions at the different sampling locations;
- Comparison of the results with previous field test results from the site (i.e., are there any similarities?); and
- Comparison of the results for a particular parameter with those from each sampling location (i.e., are the concentrations similar and do they show the same relationship for a material balance?).

In addition, the information collected for a test series is reviewed to determine if the following questions can be answered.

- Is there sufficient documentation of all procedures used in the data collection effort to allow the repetition of the effort by a person or team with technical qualifications similar to those of the original data collector?
- Is there sufficient documentation to verify that the procedures specified for use in the data collection effort were properly followed?
- Is sufficient information presented to enable potential users to evaluate the limitations of the data to determine whether the intended use of the data is appropriate?

The results of the data quality audit is used to determine whether the data collection efforts need modification and whether the use and documentation of quality control procedures are adequate.

QA/QC EVALUATION REPORTING

For QA/QC programs involving external QA/QC, the documentation of QA/QC results has been accomplished using three types of formats:

- On-site QA/QC reports to management summarizing performance audits, technical system audits, and QA problems and corrective actions;
- A letter QA report summarizing QA/QC for the field activities; and
- A quality control/quality assurance evaluation report fully documenting all QA/QC activities.

The on-site QA/QC reports advise program management of the status of critical measurement systems and data quality and may aid in quicker resolution of QA problems.

The letter QA report incorporates a compilation of all on-site QA/QC reports and a summary discussion of significant QA/QC conclusions.

The QA/QC evaluation report fully documents all QA/QC activities performed and the results of these activities. It also includes a discussion of the quality of the data generated in terms of precision, accuracy, completeness, representativeness, and comparability. It describes any limitations or constraints on the use or applicability of the data. All results of technical systems and performance audits are included, as well as the identification of any significant QA/QC problems encountered and the resolution of all corrective actions taken.

The QA/QC evaluation report substantiates the data quality achieved in the testing program and provides sufficient information to enable the user of the technical results to have confidence in his/her use of the data. It documents and explains any differences in the quality of the achieved data to the original data quality objectives stated at the outset of the test program, and states any limitations on the use of the data. A discussion of and the results of all internal audits are provided that include the effect on data quality that may be indicated by audit results not meeting the stated criteria. The table of contents for a typical QA/QC evaluation report is shown in Figure 3.

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Figure 1. Outline of typical MWC test plan.

TSA FOR HEXAVALENT CHROMIUM SAMPLING

==

Date ______ Time ______ Operator ______ Observer ______

==

Equipment identity matches pretest calibration list? ______
Probe Nozzle: stainless steel ______ glass ______
Button-hook ______ elbow ______ size ______
Cleaned according to sampling protocol? ______
Probe liner: borosilicate ______ quartz ______ other ______
Cleaned according to sampling protocol? ______
Probe heating system: ______
Checked? ______ Temperature ______ Stable? ______
Pitot tube: Type S ______ Other ______
Properly attached to probe (no interference to nozzle)? ______
Modifications: ______
Pitot tube coefficient ______
Differential Pressure Gauge: Inclined manometers ______
Magnehelics ______ Ranges ______ ______ ______
Filter Holder: borosilicate glass ______ other ______
Frit material: borosilicate glass ______ stainless ______
Teflon ______ other ______
Gasket material: silicone ______ other ______
Cleaned according to sampling protocol? ______
Filter type(s): ______
Impinger Train: number of impingers ______
Cleaned according to sampling protocol? ______
Contents: 1st ______ 2nd ______ 3rd ______
4th ______ 5th ______ 6th ______
Impinger weights recorded? ______
Cooling System ______
Proper connections? ______
Modifications ______
Silica gel: type ______ new? ______ used? ______
Barometer: mercury ______ aneroid ______ other ______
Gas Density Determination: temperature sensor ______
pressure gauge ______
Temperature sensor properly attached to probe? ______

Procedure

Recent Calibrations: pitot tubes ______
meter box ______ thermocouples/thermometers ______
Filters checked visually for irregularities? ______
Filters properly centered? ______ labeled? ______
Nozzle size properly selected? ______
All openings of sampling train plugged (pretest and posttest) ______
Impingers properly assembled? ______
Pitot lines checked for leaks and plugging? ______
Meter box leveled? ______ Manometers zeroed? ______
delta H from most recent calibration ______
Nomograph setup correctly? ______ K factor ______
Pretest leak check conducted? ______ Leakage rate? ______
Care taken to avoid scraping nipple or stack wall? ______

(continued)

TSA FOR HEXAVALENT CHROMIUM SAMPLING (continued)

==

Effective seal around probe when in-stack? ______
Probe moved to traverse points at proper time? ______
Leak check conducted during port change? before ______ after ______
Nozzle and pitot tubes kept parallel to stack at all times? ______
Filter(s) changed during run? ______
Any particulate lost during filter change? ______
Data forms completed and data recorded properly? ______
Nomograph setting changed with significant change in stack temperature? ______
Velocity pressure and orifice pressure recorded accurately? ______
Posttest leak check conducted? ______ Leakage rate ______
at inches of mercury ______
Orsat analysis? ______ Stack ______ Integrated ______
Approximate stack temperature ______ gas sample volume ______
Percent isokinetic calculated ______

Sample Recovery

==

Date ______ Time ______ Technician ______ Observer ______

==

Brushes: nylon bristle ______ other ______
Cleaned according to sampling protocol? ______
Wash bottles: glass ______ polyethylene ______ other ______
Cleaned according to sampling protocol? ______
Storage containers: borosilicate glass ______ other ______
Cleaned according to sampling protocol? ______
Cap material ______ Leak free? ______
Petri dishes: borosilicate glass ______ other ______
Cleaned according to sampling protocol? ______
Graduated cylinder: borosilicate glass ______ other ______
Subdivisions of graduated cylinder ≤2 ml? ______
Cleaned according to sampling protocol? ______
Balance type ______ Calibrated? ______
Probe allowed to cool sufficiently? ______
Probe and sample train openings covered? ______
Clean-up area(s) used ______
Silica gel: color? ______ condition? ______
Probe handling: Acetone rinse ______ other rinses ______
Particulate recovery: probe nozzle ______ probe fitting ______
probe liner ______ front half of filter holder ______
Blanks collected: reagent(s) ______
distilled water ______
Impinger rinses: distilled water ______ Other rinses ______
Recovery efficiency samples collected from this train? ______
Sample numbers: ______
______
Samples labeled and stored properly? ______
Liquid levels marked? ______
Filter handling: tweezers used? ______ surgical gloves? ______
Any particulate lost? ______

Figure 2. Example of a technical systems audit data sheet.

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Figure 3. Table of contents for an external QA/QC evaluation report.

TABLE 1. TYPICAL PRECISION, ACCURACY, AND COMPLETENESS OBJECTIVES FOR MWC TESTING

Parameter	Precision[b]	Accuracy[a]	Completeness[c]
Velocity/Volumetric Flow Rate (EPA Methods 1 and 2)	6%	10%	90 to 100%
Fixed Gases/Molecular Weight (EPA Method 3)	10%[c]	20%	90 to 100%
Moisture (EPA Method 4)	20%[d]	10%[d]	90 to 100%
Flue Gas Temperature (Thermocouple)	±10%°F	±20%°F	90 to 100%
Particulate Mass	10%	12%	90 to 100%
Flue Gas Metals (ICAP, AAS)	20%	ND	90 to 100%
Flue Gas HCl (IC Analysis)	10%	10 to 15%	90 to 100%
Flue Gas Dioxin/Furans (Modified Method 5)	50%[e]	50%[d]	90 to 100%
Flue Gas SO_2, CO, CO_2, O_2, THC (Instrumental)	20%[c]	20%[d]	90 to 100%
Flue Gas HCl (Instrumental)	25%	25%	90 to 100%
Ash (Metals)	20%	ND[f]	90 to 100%
Ash (Dioxins/Furans)	50%[e]	50%[d]	90 to 100%

[a] Precision and accuracy estimated based on results of EPA collaborative tests.

[b] Valid data percentage of total tests conducted.

[c] Coefficient of variation (CV) determined from daily analysis of a control sample, where:

$$CV = \frac{\text{Standard Deviation}}{\text{Mean}} \times 100\%$$

[d] Relative error (%) derived from audit analysis, where:

$$\text{Relative Error} = \frac{\text{Measured Value - Actual Value}}{\text{Actual Value}} \times 100\%$$

[e] Percent difference for duplicate analyses, where:

$$\text{Difference} = \frac{\text{First Value - Second Value}}{0.5\ (\text{First + Second Values})} \times 100\%$$

[f] ND = not determined for this method.

TABLE 2. SOME TYPICAL ACCEPTANCE CRITERIA, CONTROL LIMITS, AND CORRECTIVE ACTIONS FOR MWC TESTING

Problem/Procedure	Control Limits[a,b]	Corrective Action
Method 3		
Flexible Bag Leaks	None	Replace bag
Train Leaks	None; 10 in. Hg vacuum stable for $\geq$ 30 sec	Replace items as necessary
Analyzer Leaks	$\leq$ 0.2 ml for 4 minutes	Check connections
Triplicate Analysis for CO_2	Any three analyses differ by $\leq$ 0.3% absolute (CO_2 > 4%) or $\leq$ 0.2% absolute ($CO_2 \leq$ 4%)	Continue to repeat analysis until control limit is met
Triplicate Analysis for O_2	Any three analyses differ by $\leq$ 0.2% ($O_2 \geq$ 15%) or $\leq$ 0.3% (O_2 < 15%)	Continue to repeat analysis until control limit is met
Manual Sampling		
Flow Alignment	Avg. resultant angle within 20°	Adjust flow data to correct for misalignment based on misalignment angles
Isokinetics	100 $\pm$ 10%	Qualify data
Final Leakrate (after each port)	$\leq$ 0.02 acfm or 4% of sampling rate, whichever is less	Adjust sample volume for port
Dry Gas Meter Calibration	Post average factor (Y) agees $\pm$ 5% of pre-factor	Adjust sample volumes using the Y that gives smallest volume
Individual Correction Factor (Y_i)	Agree within 2% of average factor	Redo correction factor
Intermediate Dry Gas Meter	Calibrated every six months against EPA standard	Recalibrate against EPA standard
Analytical Balance	0.1 g of NBS Class S Weights	Repair balance and recalibrate
Barometric Pressure	Within 2.5 mm Hg of a mercury-in-glass barometer	Recalibrate
CDD/CDF Analysis		
Internal Standard Recoveries	100 $\pm$ 50%	Re-extract and reanalyze
Surrogate Recoveries	100 $\pm$ 50%	No action
Verification of Identification		
1) Ratio of M+ to M+2 or M+2 to M+4	Within 20% of theoretical value, except for TCDD/TCDF which are taken within 13%	Reconsider peak identification
2) Retention Time	Within 3 seconds of the corresponding or nearest ^{13}C internal standard or surrogate standard (with reference to continuing calibration)	Reconsider peak identification
3) Signal-to-Noise Ratio	Greater than 2.5	Reconsider peak identification

(continued)

TABLE 2. (continued)

Problem/Procedure	Control Limits[a,b]	Corrective Action
Proof Rinse Blanks	Less than target limits of detection	Determine source of contamination; change reagents and/or reclean glassware
Reagent Blanks (if analyzed)	Less than target limits of detection	Change reagents
Field Blanks	Less than target limits of detection	Determine source of contamination; change reagents and/or reclean glassware
Audit Sample(s)	Relative error $\leq$ 50%	Reanalyze
Replicates	Difference $\leq$ 50%	Reanalyze
CEM Measurements		
Linearity Multipoint Calibration (four points)	$R \leq 0.9950$	Adjust instrument, redo multipoint
Daily Drift (zero and span)	a) $< 5\%$ of span b) $\geq 5\% < 20\%$ of span c) $> 20\%$ of span d) 2 days with drift greater than 10% indicates instrument maintenance	Data not adjusted Adjust assuming linear drift Reject data
Sampling System Bias	± 5% of span	Check heat tracing and/or clean sample line
Daily QA Check (mid-range)	± 10% of certified concentration	Redo initial calibration
Response Time	Less than one minute	Increase sample flowrate
Interference Check	± 7% of manual result	Repeat interference check to verify; if verifed, clean sample lines and check calibration
Stratification Check	Each point with 10% of average of all points	Use point of average concentration
Line Leak Check	0.5% O_2	Locate and repair leak, recheck
Manifold Leak Check	0.5% O_2	Locate and repair leak, recheck

[a] Relative error (%) derived from audit analyses, where:

$$\text{Relative Error} = \frac{\text{Measured Value} - \text{Actual Value}}{\text{Actual Value}} \times 100\%$$

[b] Difference (%) for duplicate analyses, where:

$$\text{Difference} = \frac{\text{First Value} - \text{Second Value}}{0.5\ (\text{First} + \text{Second Values})} \times 100\%$$

TABLE 3. TYPICAL AUDIT MATERIALS FOR MWC EXTERNAL QA/QC PROGRAM

Type of Audit Material	Range of Audit	Source of Audit Material	Acceptance Criteria
		General Meter Box Audits	
Temperature Devices	NA[1]	ASTM[2]	Temperature measuring devices ± 1.5% (abs.)
	NA	ASTM	
Critical Orifice	NA	EPA-QAD[3]	Dry gas meters ± 5%
	NA	EPA-QAD	
		Continuous Emission Monitoring System Audits	
NO	100 ppm_v	Protocol 1[4]	± 10% of audit gas
O_2	9.0%	Protocol 1	± 10% of audit gas
CO_2	11.0%	Protocol 1	± 10% of audit gas
CO	450 ppm_v	Protocol 1	± 10% of audit gas
SO_2	250 ppm_v	Protocol 1	± 10% of audit gas
	100 ppm_v	Protocol 1	± 10% of audit gas
	50 ppm_v	Protocol 1	± 10% of audit gas
THC	10 ppm_v	EPA QAD	± 10% of audit gas
HCl[5]			
		Manual Sampling and Analysis Audits	
Particulate[6]	50 mg	EPA QAD	± 10% of audit value
Particulate	200 mg	EPA QAD	± 10% of audit value
Toxic Metals: Filter Matrix	High: 50 to 2500 ug	EPA QAD	± 10% of audit value
	Low: 25 to 1250 ug	EPA QAD	± 10% of audit value
Mercury	20 ug/ml	EPA QAD	± 10% of audit value
Mercury	5 ug/ml	EPA QAD	± 10% of audit value
Hexavalent Cr	4 ug/ml	EPA QAD	± 10% of audit value
CDD/CDF	Low	EPA QAD	Within 50% of 90% CI
	Medium	EPA QAD	Within 50% of 90% CI
	High	EPA QAD	Within 50% of 90% CI
VOST	ppb	EPA QAD	± 50% of audit value

[1] Not Applicable.

[2] American Society for Testing and Materials.

[3] Environmental Protection Agency, Quality Assurance Division.

[4] Certified audit gas meeting EPA Traceability Protocol 1.

[5] Accuracy of the HCl CEMS's are verified by the results of manual HCl sampling.

[6] Audit device sampled and particulate catch analyzed once during performance test.

DEVELOPMENT AND IMPLEMENTATION OF EPA DIOXIN/FURAN EMISSION TESTING PERFORMANCE AUDIT PROGRAM

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ABSTRACT

The Quality Assurance Division (QAD) of the Atmospheric Research and Exposure Assessment Laboratory of the U. S. EPA is responsible for developing performance audit materials and/or devices for assessing stationary source emission measurements. One of the latest additions to the QAD audit program is performance audit samples for the analysis of dioxin/furan emission samples. This paper describes (1) development of the dioxin/furan audit material, (2) establishment of the audit values for each isomer in the audit samples, (3) distribution and use of the audit samples, and (4) calculation, interpretation, and reporting of the audit results using QAD's computer program.

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INTRODUCTION

In 1977, Olie et al. reported the occurrence of polychlorinated dibenzo-p-dioxins (CDDs) and polychlorinated dibenzofurans (CDFs) in fly ash samples from three municipal waste combustors in the Netherlands.[1] Further work confirmed the presence of CDDs/CDFs in nanogram per gram (ng/g or ppb) to picogram per gram (pg/g or ppt) levels in fly ash and ng to pg amounts in air source emissions of municipal waste combustors, and in addition, in the air emissions and fly ash of many other incineration sources.[2]

Other research has shown that 75 CDD and 135 CDF isomers exist with a number posing a significant health risk.[2] Toxicities of specific isomers and isomer groups vary by a factor of 100,000. The isomers with 4 to 6 chlorine atoms substituted on the 2,3,7, and 8 positions of the benzene rings are the most toxic; however, they usually account for only a small percentage of the total CDD/CDF.

Concern with the potential health risk posed by the CDD/CDF within the emissions from various combustion sources has resulted in a substantial increase in CDD/CDF emission sampling and analysis in the last several years. To differentiate the isomers which contribute to the health risks and to measure the isomers at the levels at which they are present (pg or ng per dry standard cubic meter of air), requires extremely sensitive and specific analytical procedures for isolation, identification, and quantification of the compounds. The generally accepted isomer-specific analytical procedure involves gas chromatography/mass spectroscopy (GC/MS) in the "multiple ion selection" mode. The instrument is calibrated using standards containing native and isotopically-labeled CDD/CDF isomers representative of the various isomer groups (e.g., tetra-CDD, penta-CDF, etc.). Quantification of the specific isomers and isomer groups in the sample is based on isotopically-labeled CDD/CDF internal standards which are added to each sample prior to the sample extraction/cleanup procedures.

Because the GC/MS analysis is very complex, time consuming, and expensive, and the quality control criteria for the sample preparation and analysis are many and complex, the EPA decided that a relatively simple means to assess the CDD/CDF data from the increasing number of laboratories involved in this analysis was needed. It was concluded that a performance audit material containing the CDD/CDF isomers of interest in a suitable matrix would allow the determination and documentation of laboratory capability to extract, cleanup, and analyze stack gas emission samples. In addition, development of a CDD/CDF audit material would be consistent with the emission testing audit program already being administered by the Quality Assurance Division (QAD) of EPA's Atmospheric Research and Exposure Assessment Laboratory (AREAL).

DEVELOPMENT OF AUDIT MATERIAL

Development of an audit material for the analytical portion of CDD/CDF emission testing was initiated in December 1986. XAD-2[a] resin was selected as the audit sample matrix because it is routinely used for sample collection in the modified Method 5 train during CDD/CDF emission testing. Previous development of dioxin performance evaluation samples in soil and adipose tissue matrices had demonstrated that it was possible to produce a homogeneous audit sample mixture.[3]

To start, 12 kg of XAD-2 resin were purchased, cleaned by extraction, and checked for residual organics. The resin was divided into a number of batches which were spiked with various levels of CDD/CDF typically found in emission samples from incinerators and other combustion processes. The spiking was performed by EPA's Environmental Chemistry Laboratory (ECL) in Bay St. Louis, Mississippi, which had been responsible for preparation of the soil and adipose matrix samples. The spiking samples consisted of native CDD/CDF's extracted from fly ash along with specific selected tetra- through octa-isomers; this resulted in a distribution of CDD/CDF isomers typical of those in combustion emissions. Following spiking, the 40 g samples of resin were packed in amber colored glass bottles and stored under refrigeration.

To determine and confirm the anticipated CDD/CDF concentrations at the various spike levels and to verify the homogeneity of the audit samples, three samples from each level were subjected to isomer-specific analyses by the EPA CDD/CDF laboratory in Research Triangle Park, North Carolina. Following this, one sample at each level was analyzed by each of two additional laboratories (Wright State University and the New York State Department of Health) to provide supplemental data for development of confidence intervals for use in audit data assessment. The remainder of the samples were transferred to QAD for storage and distribution.

DISTRIBUTION OF SAMPLES

The CDD/CDF audit samples are now available upon request by Federal, State, and local agency staff for use in agency-administered CDD/CDF emission testing programs. The contact is: Mr. Thomas J. Logan, Source Branch (MD-77A), Quality Assurance Division, Atmospheric Research and Exposure Assessment Laboratory, Research Triangle Park, NC 27711.

Upon request, two audit samples (three, if inlet and outlet sampling is to be conducted) will be shipped to the requesting agency along with auditee/auditor instructions. If directed, the audit samples can be shipped directly to the analytical laboratory. This may be preferable since the samples contain ng quantities of CDD/CDF and are considered hazardous for shipping purposes. Under D.O.T. regulations, they require special packaging and may be transported only on "cargo-only" aircraft. If received by the agency auditor, the audit sample package should only be opened to the point needed to remove a copy of the auditee/auditor instructions; only the analytical laboratory should open inner sample packaging. If samples are sent directly to the analytical laboratory, a copy of the auditee/auditor instructions will be forwarded to the agency auditor. The audit samples should be stored under refrigeration whenever possible during the period between when they are received and analyzed.

AUDIT SAMPLE ANALYSIS AND RESULTS REPORTING

Upon delivery of the audit sample package, the analytical laboratory representative (auditee) should verify receipt of the two (or three) samples and instructions and check for damage to the samples. The audit samples should be analyzed at the same time and in a manner identical to the field samples. If a confirmational analysis (when analysis on one or more additional columns is used for further separation and quantification of specific isomers) is used for the field samples, it should be used for the audit samples. The auditee is responsible for completing the audit data form (see Figure 1) for each sample. The values recorded must correspond directly to the specific isomers and isomer groups shown. Results cannot be reported in any other format, as they will be input directly from the standardized form into the EPA/QAD computer program for calculation and reporting; the EPA does not interpret other data formats. The data format shown in Figure 1 was selected to allow calculation of "toxic equivalents" for the various isomers and isomer groups. If an isomer or isomer group is not detected, the auditee enters "ND" in the appropriate space and indicates the detection limit in parentheses.

The auditee then forwards the results to the agency auditor who in turn forwards them to QAD for processing. It is recommended that the auditee retain a copy of the form to aid in resolving any questions which may arise during the audit data analysis.

AUDIT RESULTS CALCULATION AND INTERPRETATION

Because of complexity of the CDD/CDF analysis and the inability to provide audit samples of exactly known and certified concentrations, the confidence intervals from previous analyses of the audit sample levels are used in assessing the audit data. QAD has developed a computer program which facilitates comparison of audit results for the individual CDD/CDF isomers and isomer groups to the established confidence intervals.

To develop this program, a 90% confidence interval was calculated for each of the 25 CDD/CDF isomers and isomer groups using the values reported by several laboratories. If an isomer or isomer group was not detected, three times the reported detection limit was used in the confidence interval calculation (a parallel calculation is used in the assessment of audit results). The average values for the specific CDD and CDF from each of the different audit sample concentrations analyzed were used to determine the confidence intervals for most isomers. However, in cases where the reported values for an isomer showed an actual interval significantly larger than the average interval, the actual confidence interval was used. The interval based on the average of the values was used because the individual 90% confidence intervals for the isomers varied greatly due to the small number of laboratories analyzing the audit samples. As data from the audited laboratories are obtained, the confidence intervals will be periodically recalculated and updated.

To evaluate a set of audit data, the isomer and isomer group values and other pertinent audit information, including the sample identification number, are input directly from the form in Figure 1 as prompted by an identical form in QAD's dioxin audit sample computer program.

For each of the 11 CDD entries and the 14 CDF entries, the audit value is compared to the appropriate 90% confidence interval. The number of values outside this interval are counted and presented in a format similar to that shown in Figure 1, Items 1) and 3). An additional range, 50% above and 50% below the 90% confidence interval, has also been calculated. The bottom limit of this range is the value 50% of the lower value of the 90% confidence interval. The upper limit of the range is the value 150% of the upper value of the 90% confidence interval. For example, if the 90% confidence interval covers the range from 0.4 to 2.0 ng, the range 50% above and 50% below the 90% confidence interval extends from 0.2 to 3.0 ng. To determine the number of values outside the 90% confidence interval and within 50% of the confidence interval, the values reported in Items 1) and 3) can be subtracted from the values reported in Items 2) and 4), respectively.

In an effort to provide an estimate of the impact that the audit analytical results would have in relation to the toxicity of specific CDD/CDF isomers, the percent errors for the analytical values outside the 90% confidence interval are multiplied by the appropriate 2,3,7,8-TCDD toxic equivalency factors, and the average weighted absolute percent error is calculated. The bias is then calculated using the individual weighted percent errors. If the bias is positive (+), the auditee's analytical values are higher than the true values. If the bias is negative (-), the auditee's weighted values are lower than the true values. The average percent error and bias are not intended to be used for direct correction of the emission data, because (1) the number of audit samples analyzed by the auditee is usually too small and (2) the ratio(s) of the amounts of CDD and CDF isomers in the audit sample(s) may be different from those in the emission samples.

Following these calculations, the program generates a hard copy audit report for each audit sample. A copy of this report is forwarded to the agency auditor. It does not include information on which isomers or isomer groups were not within the confidence intervals. If required, the auditor may obtain this information by contacting QAD. However, this information is never released to an auditee. QAD does not determine the acceptability of an audit result. This decision is left to the requesting agency based on the reported confidence interval results and "toxic equivalency" based bias. As previously mentioned, the 90% confidence intervals will be updated as audit data are amassed. QAD also plans to conduct a study in the near future to assess stability of audit samples.

REFERENCES

1. K. Olie, P. G. Vermeulen, and O. Hutzinger, Chemosphere, 6: 455 (1977).

2. C. Rappe, Environ. Sci. and Tech., 18(3): 78A (1984).

3. R. K. Robeson, "Preparation and Analysis of Dioxin Performance Evaluation Samples," EPA Report No. 600/X-83-053, August 1983.

AUDITOR ____________ AGENCY ____________
AGENCY ADDRESS ____________ AGENCY PHONE NO. (   ) ____________
AUDITEE COMPANY ____________ ADDRESS ____________
AUDIT SAMPLE NO. ____________ DATE AUDIT SAMPLE RECEIVED ____________
DATE ANALYZED ____________ CONFIRMATION ANALYSIS USED: YES __ NO __
AUDITEE'S NAME ____________ SIGNATURE ____________

Compound	Auditee Result (ng/sample)	Compound	Auditee Result (ng/sample)
2378-TCDD		2378-TCDF	
Other TCDD		Other TCDF	
123789-PCDD		12378-PCDF	
Other PCDD		23478-PCDF	
123478-HxCDD		Other PCDF	
123678-HxCDD		123478-HxCDF	
123789-HxCDD		123678-HxCDF	
Other HxCDD		123789-HxCDF	
1234678-HpCDD		234678-HxCDF	
Other HpCDD		Other HxCDF	
OCDD		1234678-HpCDF	
		1234789-HpCDF	
		Other HpCDF	
		OCDF	

The results of the above performance audit sample will be calculated as follows:

RESULTS OF THE POLYCHLORINATED DIBENZODIOXINS (CDD)

1) ** of the 11 different CDD are not within the 90% confidence intervals.
2) ** of the 11 different CDD are not within 50% of the 90% confidence intervals.

RESULTS OF THE POLYCHLORINATED DIBENZOFURANS (CDF)

3) ** of the 14 different CDF are not within the 90% confidence intervals.
4) ** of the 14 different CDF are not within 50% of the 90% confidence intervals.

RESULTS BASED ON A 2,3,7,8-TCDD TOXIC EQUIVALENCY FACTORS

5) Based on the 2,3,7,8-TCDD toxic equivalency factors, the average percent error outside the 90% confidence limits was **% with an average of **%.

(**) - Represents values calculated by the U. S. EPA.

Note: For those CDD and CDF reported as not detected by the auditee, three times the reported detection limit (in parentheses) will be used for the calculations. The identical procedure was used for calculating the confidence intervals.

Figure 1. Reporting form for U. S. EPA Quality Assurance Division dioxin/furan performance audit sample data.

TOTAL QUALITY: AN APPROACH TO MORE EFFECTIVE MWC DECISION-MAKING

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ABSTRACT

The incineration and subsequent ash disposal of municipal solid waste (MSW) pose many technical, economic, regulatory, and social issues for decision-makers in the private and public sectors. These issues include the design of more efficient incinerators, adequate characterization of the waste (gas and ash), and selection of emission control systems and monitoring systems. Consequently, effective MSW management poses a challenge from many complex variables which decision-makers must consider. Experience demonstrated in business and, more recently, in government suggests that a Total Quality approach to MSW management may yield significant value and benefits to the decision-makers who must plan, implement, and evaluate MSW options. Recent work in EPA involving the application of Total Quality Management (TQM) to environmental data operations indicates promise of improved quality and productivity to decision-makers who require environmental data as an integral part of their decisions.

The TQM approach focuses the efforts of managers and workers toward understanding and fulfilling the needs of the customer, renewed and effective communications across all levels of the organization, and reliance on quality standards and measures of of effectiveness in order to achieve the desired goals successfully. Key elements of this approach include upfront planning (to define clearly what products or services and quality are needed and how the products or services are to be used), effective implementation of the planned QA/QC systems (to develop and document a logical and defensible rationale for the decision to be made), and review and evaluation of process (to measure progress and assure that expectations are being met). The application of this approach to the use of enviromental data for municipal waste combustion (MWC) decisions will provide decision-makers with improved quality, efficiency of operations, cost-effectiveness, and public confidence.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for presentation and publication.

PRINCIPLES OF TOTAL QUALITY MANAGEMENT

WHAT IS TOTAL QUALITY MANAGEMENT

Total Quality Management (TQM) is a process involving a management philosophy and operating methodology completely committed to quality improvement in all aspects of the operation or program. TQM is based on the fundamental concept of "doing the right thing the right way and doing it right the first time." TQM focuses on continuous process improvement. Each activity is viewed as a process, whether it produces a product or a service. Through the use of scientific methods, such as Statistical Quality Control (SQC), improvements to the product or service may be achieved by changing the process by which it is made or delivered. The expectations of the customer become the measure of success. Perfection is the goal. The "customer" is the one who receives the product or service, whether he is internal or external. If his needs and expectations are met, then success has been achieved.

The concept of TQM seems quite logical to most people, yet it has been slow to take hold in business and government in the United States. It has largely been the dramatic economic advances of Japan which has "shocked" many American businesses into action. TQM has been institutionalized in Japan since 1949 and has been credited as the primary force behind Japan's growth in the world marketplace in the past 30 years. In the 1950s, "made in Japan" meant "cheap." Now, in the 1980s, "made in Japan" means Quality. The irony is that TQM is not a Japanese concept. It originated here in the United States and was brought to Japan in the late 1940s and early 1950s by distinguished Quality experts like Dr. W. Edwards Deming and Dr. Joseph M. Juran. Japan embraced the concept of TQM and the result is history. To foster and promote TQM in Japan, a national Quality award was created in the early 1950s to recognize achievments in Quality. The competition among Japanese companies for this annual award is very intense today as it was at the outset. Again with irony, the award for Quality in Japan is named for an American: the Deming Prize.

American industry is responding to the Japanese Quality challenge. TQM has become rooted in many facets of American industry. More important is the recognition that TQM does not apply only to business or manufacturing, but applies to any and all activities undertaken by government and the private sector. The elements of TQM are straight-forward and embrace a common-sense approach to management. These are:

- A Focus on the Customer
- A Long-term Commitment
- Top Management Support and Direction
- Employee Involvement

- Effective and Renewed Communications
- Reliance on Standards and Measures
- Commitment to Training
- Importance of Rewards and Recognition

Focus on the Customer

Every functional unit has a customer. As noted above, the customer may be internal to the organization, such as another department or group, or external to the organization. TQM advocates that managers and employees become so customer focused that they continually find new and innovative ways to meet or exceed the customer's expectations. Fundamental questions which should be asked include:

- Who are my customers?
- What do they need from me?
- What is my product or service to meet their need?
- What are their measures, their expectations?

Long-term Commitment

TQM does not happen overnight. Experience in business and in government shows that substantial gains occur only after management makes a long-term commitment to improving Quality. Often, this commitment must be maintained for five years or more before full value and benefits are achieved. Throughout this process, customer-focus must be renewed constantly to keep that goal foremost.

Top Management Support and Direction

Top management must be the driving force behind TQM if it is to succeed. Senior managers must exhibit personal support by using Quality improvement concepts in their management style, by incorporating Quality in their strategic planning process, and by providing financial and staff support.

Employee Involvement

Full participation by employees at all levels of the organization is also an integral part of the TQM process. Each employee must be a partner in achieving Quality goals. Teamwork involves managers, supervisors, and employees in improving service delivery, solving systemic problems, and correcting errors wherever they occur in the work processes.

Effective and Renewed Communications

The power of internal communications, both vertical and horizontal within the organization, is central to employee involvement. Regular and meaningful communications among all levels must occur. This will allow the organization to adjust its ways of operation and reinforce the commitment to TQM at the same time. Experience in business has shown that some of the best ideas for improving efficiency and productivity come from those employees who are closest to the work on a daily basis. Open channels of communication assure that problems are identified without recrimination and that solutions are found quickly and are most effective.

Reliance on Standards and Measures

In order to achieve a goal, there must be some measure to enable one to know when the goal has been attained. Measurement is the springboard to involvement. Measurement allows the organization to initiate corrective action, set priorities, and evaluate progress. The standards and measures should reflect the needs and expectations of the customer, and changes that need to be introduced in the internal business of providing those needs. The emphasis is on "doing the right thing right the first time." Three more fundamental questions may be added to the four given earlier. They are:

- What is my process for meeting the customer's need?
- Does my product or service meet or exceed their need and expectations?
- What action is required to improve my process?

Commitment to Training

Training is absolutely vital to the success of TQM. Managers and employees alike must be given the tools needed in order to bring about the changes necessary for TQM to flourish. TQM usually begins with "awareness" training for for teams of top level managers. This is followed by courses for teams of mid-level managers and supervisors, and finally by courses for non-managers. Awareness training is followed by an identification of areas of concentration, or of functional areas where TQM should be introduced first. Just as Quality improvement is continual, so is training. As managers and employees grow in their knowledge and application of TQM, their skills must be continually increased to enable the improvement process to continue.

Importance of Rewards and Recognition

People like to be told that they are doing a good job. TQM includes the recognition of achievements and good performance to demonstrate to employees that they are important to the organization and that their contribution is appreciated. It is perhaps surprising that successful TQM programs have not had to rely on financial rewards as incentives for Quality improvements, but recognition of individuals or groups among their peers has been most effective in fostering pride and ownership of achievements.

HOW DOES TQM WORK

TQM stresses the development and evolution of a "Quality Culture" in the organization. If management accepts ownership of a cultural change in their organization and uses the change brought by TQM effectively, improvements will occur. TQM means working together (management and employees) to resolve problems in an environment of mutual responsibility. Finding the solution to the problem is the primary objective, not finding out whose fault it was. This helps to eliminate finger-pointing when problems arise and which frequently keeps the real causes of problems from being understood. Instead, TQM requires that someone take ownership of the problem and solve it. The key to TQM institutionalization lies in management commitment. If management, who are usually the decision-makers, drives the process and incorporates the philosophy of TQM into their management styles, improvement will result in the operation and in the products or services produced.

Since TQM requires total commitment, participation and recognition at all levels of an organization are key to its success. This means that there must be open communications among all levels to assure that the problems and their solutions are fully understood. This may require the formation of new partnerships among organizational units that may rarely have communicated before. Very often, quality and efficiency have suffered because of a lack of communication and understanding among different work units within an organization. Through improved communications, different work units develop their own customer-client relationships and, through TQM, develop an increased awareness of and sensitivity to customer needs.

Total Quality Management sounds simple. It is not easy to implement, but the rewards of increased Quality and productivity have been demonstrated. What began in Japan forty years ago by Deming and others has returned to this country and gained growing acceptance in business and in government. Fortune 500 companies which have adopted and implemented TQM include Westinghouse, Corning Glass, Hewlett-Packard, Ford Motor Company, and many others. In every case, the focus has been on **"doing the right thing right the first time."**

The issues affecting Municipal Waste Combustion may not, at first, seem to be appropriate for Total Quality Management. However, in all aspects of MWC design, construction, and operation, there are customers whose needs must be satisfied. Before examining some specific ways in which TQM can improve MWC management, it is appropriate to look at some of the issues facing decision-makers for MWC facilities.

MWC DECISIONS/DECISION-MAKING

WHAT ARE THE ISSUES

Municipal Waste Combustion (MWC) is growing in importance as a method for disposing municipal solid waste (MSW), particularly as landfill disposal becomes less available to municipalities faced with growing waste loads. MWC or incineration has historically been used frequently as a means of municipal waste disposal. However, as waste loads grew in heavily populated areas and public concern for air quality increased, the use of incineration began to diminish. The promulgation of air quality standards in the late 1960s and early 1970s further reduced the viability of incineration for MSW disposal. However, even before the emergence of air quality regulations began to limit the use of incineration, the growth of waste production had led most municipalities to use landfills as the principal means of MSW disposal. Now, these same municipalities are running out of landfill space and MWC has emerged as a possible waste disposal option being considered by many cities and towns.

The issues associated with MWC involve technical design, adherence to regulations, the cost-effectiveness of MWC as a disposal option, and the perception by the general public of its health and safety impacts. Advances in research and engineering have produced significant improvements in the design of combustion units which have yielded greater efficiency and reliability in combustor operation. These advances have helped to make MWC a more cost-effective alternative, even considering the capital cost requirements of the MWC plant and its needed environmental controls. Adherence to environmental regulations must be demonstrated by the operating plant. The effectiveness of the environmental control systems to meet emissions and effluents regulations may significantly bear upon the public acceptance of the MWC plant. Of course, in some cases, even the most pristine MWC operation will not always allay the fears and objections of some segments of the community. Opponents who take a "Not in my backyard" stance may not ever be convinced that the MWC facility poses no health threat even though all applicable environmental standards may have been met or exceeded.

WHAT ARE THE KEY DECISIONS

Important decisions occur for decision-makers during MWC facility design, construction, and operation. It is assumed that Quality Assurance and Quality Control are integral components of the decision-making process.

Technical Design

Quality in the technical design is an imperative. Simply, the facility must be designed to meet the needs and expectations of the customer, the owner of the facility. Standards and regulations will require that the facility be able to perform satisfactorily. Everything that goes into the design of the systems comprising the MWC facility, from waste handling to emissions controls, has customer needs and expectations defined. The facility must be able to operate safely and to handle a specified waste load with a prescribed efficiency in order to be a cost-effective waste disposal alternative. Provisions must be made for ash disposal that meet all applicable regulations.

Construction

During construction of the facility, the technical design establishes the standard of Quality for all activities. Decision-makers must deal with suppliers of equipment and material, whose products may not always meet the quality specifications. In some cases, shoddy construction may result in costly rework that could have been avoided if the right work had been done correctly the first time.

Operation

The operation of a MWC facility must be in accordance with many regulations. Decision-makers must have confidence in the monitoring data to be sure that the facility operates within the limits of environmental standards. In addition, the cummulative Quality gained through design and construction is tested through the operation of the facility to determine if the customer's needs and expectations are met. That is, does the facility operate as it was intended.

TQM IN MWC DECISION-MAKING

Total Quality Management may be applied to every facet of MWC design, construction, and operation. The principles of TQM are independent of the type of work being performed. While it is possible to show in detail how TQM may be applied to all aspects of MWC facilities, such a discussion is beyond the scope of this paper. Instead, it is more illustrative to focus on one specific area pertaining to MWC in order to show how TQM may be applied and what value and benefits may be derived.

Consider the selection, installation, and operation by the facility owner of stack emissions monitoring equipment. TQM requires the involvement of management and employees, driven by a total commitment from management, to continual improvement of the monitoring process. TQM focuses on the needs of the customer and implements a management process designed to meet those needs. For this example, TQM may applied in the following manner:

STEP 1. WHO ARE MY CUSTOMERS?

In regard to environmental monitoring equipment, there several customer and client relationships. First, the owner is a customer for the supplier of the monitoring equipment. He must define his needs and expectations based on the performance required in order to detect and quantify emissions from the MWC facility during various types of operating conditions. The owner is also a supplier of a waste disposal service. His customers include various regulatory agencies, which require the facility to meet various standards, and the general public, who need and expect the facility to be operated safely and to pose no threat to human health or the environment.

STEP 2. WHAT DO THEY NEED FROM ME?

The needs of the owner for monitoring equipment are determined by the needs of the owner's customers for the performance of the facility. That is, the regulatory community and the general public need to be sure that the emissions from the MWC facility are within safe limits during all types of operation with different types of wastes. The decision-makers and employees will be more aware through TQM participation of these concerns and how the operation of the facility impacts the concerns. The regulations require compliance. The public may expect more, such as frequent communications to keep the public informed of the MWC operation. This may simply mean being a "good neighbor," but awareness of and sensitivity to public issues may significantly reduce public concerns, whether real or perceived.

STEP 3. WHAT IS MY PRODUCT/SERVICE TO MEET THEIR NEED?

The service provided by the owner is to monitor the stack emissions according to prescribed regulations to detect any exceedances of emission limits. If detected, the owner will take corrective action to reduce the emissions and eliminate the problem. TQM implicitly assumes that the owner has in place an effective Quality Assurance and Quality Control program to enable him to determine with confidence when emission limits have been exceeded. Such a QA/QC program should include periodic audits and performance evaluations to assure that the equipment is working properly and that the personnel operating the equipment have the necessary training in order to operate the systems according to specifications. In a TQM environment, such training would have been identified early in the planning for the facility and all operating personnel would have received the necessary instruction.

STEP 4. WHAT ARE THEIR MEASURES AND EXPECTATIONS?

Up-front planning with the owner's technical staff or consultant is essential to determine the data quality objectives for the monitoring operation; that is, what data are needed, what must be the quality (e.g., precision, bias, representativeness) of the data, and how are the data to be used in determining compliance or non-compliance. In most cases, the specific standards and measures are prescribed by the regulations. These measures provide a basis for assessing the performance of the monitoring systems. In the TQM process, the audit and performance evaluation functions provide a means for continually improving the operation of the monitoring system (i.e., greater precision, fewer system outages, etc.). Such improvement increases the confidence of decision-makers in the validity of the data obtained and, consequently, in the decisions they make.

STEP 5. WHAT IS MY PROCESS FOR MEETING THEIR NEED?

The needs of the customers will be met if a goal of "zero defects" is achieved. While there are sure to be skeptics who will argue that such a goal is unrealistic for MWC facility operation, experience from other business environments suggest otherwise. TQM has shown that when the eight elements of TQM are implemented, it is very realistic to expect and demand "zero defects." In this example, "zero defects" in the operation of monitoring equipment may be achieved through a vigorous testing and maintenance program for the equipment and constant vigilence by the operating personnel for any suspected problems. Such a process is called Quality Improvement and it is continuous. Quality Improvement may be applied using techniques like Quality Circles, in which small groups of workers meet regularly to identify and solve systemic problems. Through open and effective communications, management is kept informed of the Quality Improvement process so that more effective decisions may made regarding needed changes. By having the responsibility for "zero defects" placed close to the source of the work, problems are identified more quickly, solutions are found more readily, and costly outages of the systems are minimized.

In every situation, the keys to the process are understanding what is goinig on, having measures of process performance to assess progress or effectiveness, collecting the necessary information or data to be able to analyze the condition, and taking prompt and effective corrective action. TQM creates a management environment in which these keys are implicit in every activity. After the successful completion of a Quality Improvement effort, the cycle begins again to find new and better ways of operating the system. There is considerable evidence which shows that there is no shortage of ideas for Quality Improvements in TQM organizations.

STEP 6. DOES MY SERVICE MEET THE CUSTOMER'S NEED AND EXPECTATIONS?

In every stack monitoring activity, there are periodic checks made to assure that the facility's emissions are within prescribed limits. Such checks serve to document that needs and expectations are being met. But perhaps of greater significance is the TQM process itself which sends a message to the customers that Quality is important and that the customer's needs and concerns have been heard. Today, when so much seems to be determined by perceptions, it is clearly in the best interests of the MWC facility owner or manager not only to demonstrate a Quality operation but also to be perceived by others as doing so.

TQM brings reality to such perceptions. Continuous Quality Improvement provides observers with tangible evidence of the organizations commitment to the customers. By keeping this question foremost in their minds, decision-makers are likely to be much more effective in meeting their own goals.

STEP 7. WHAT ACTION IS NEEDED TO IMPROVE MY PROCESS?

This question results from the TQM process of continuous Quality Improvement. It is, in a sense, the product of the TQM process because the action that is required will be obvious to decision-makers. All of the principles applied earlier -- customer focus, employee involvement, effective communications, reliance on measures, and the rest -- lead to a culture within the organization in which actions necessary to improve Quality are identified early and represent the total knowledge and experience of of the organization. It reflects a decision-making philosophy that no one of us is smarter than all of us.

By having those close to the work involved in the decision-making process, the operation of emissions monitoring systems may be made more effective. Management is better informed about problems and their solutions, and may act with increased confidence that the decisions are correct. More important, perhaps, is that if mistakes are made, TQM provides a mechanism for analyzing the mistakes and attaining a quick resolution.

SUMMARY

Total Quality Management offers a new approach to management of technical operations and organizations which may, at first, seem diffcult or impractical to many. But TQM acceptance and implementation is growing in business and government throughout the country. Its track record is solid. Total Quality Management works.

To apply TQM to the management of MWC facility design, construction, and operation may also seem impractical at first. Clearly, the decision-makers involved in MWC want to do Quality work and to produce and operate facilities in a Quality manner. TQM brings a new concept to management that is simple and undeniable: do the right thing the right way the first time. Through TQM, operations will improve. Costs will be reduced in the long-term and customer satisfaction will grow. Try it.

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ABSTRACT

STABILIZATION OF HEAVY METALS IN INCINERATOR ASH AND POTENTIAL RE-USE OF THE HARDENED MATERIAL

by William Strauss, Director
Harrisburg's 720 ton-per-day Waste-to-Energy Facility

The City of Harrisburg owns and operates one of the oldest continuously operating waste-to-energy plants in the United States. The facility has been on-line since 1972. The two 360 ton-per-day Martin/International Boiler Works furnaces have processed over 2 1/2 million tons of refuse with the recovered energy suppling steam to the Bethlehem Steel Mill, the district heating system, and to the City owned 8 megawatt power generating station.

We became aware of a very recent successful de-listing of Electric Arc Furnace (EAF) dust through a proprietary patented process. The physical and chemical characteristics of EAF are very similar to the flyash produced in the combustion of refuse. The process succeeds in producing a material that chemically isolates the heavy metals from the environment. Furthermore, the final product is a solid.

We are also aware of several methods of creating a solid material from MSW ash through a melting and resolidifcation of the molten material.

The City has therefore funded a research program that will test the EAF process and the melting process on MSW incinerator flyash.

Also, there are several patents existing that describe processes for the re-use of combined incinerator ashes (bottom ash and flyash). The insurmountable hurdle however has been the potential leaching of the heavy metals. The research project will also explore the possibility of combining the processes that are to be evaluated with one or several of these patents in order to produce a material that is both environmentally benign and useful.

The project will quantify the mobility of the heavy metals and will also test the end product's strength as a potential construction material.

INTRODUCTION

The City of Harrisburg has owned and operated one of the nation's oldest waste-to-energy facilities. Operations began in 1972 and have continued until the present. On January 12, 1989, the facility reached a significant milestone with the processing of the 2 1/2 millionth tons of municipal solid waste. The facility has been the regional solution for disposal of municipal solid waste for the last 17 years.

The city has benefited in many ways as a result of the facility. Low cost energy is being supplied to the local steel mill, helping the mill to remain competitive and retain the thousands of jobs it provides. A district heating system supplemented by low cost steam from the waste-to-energy plant has helped create favorable heating costs for shops and businesses in the central city. And the city, which supplies only 12% of the waste required to fuel the operation, has enjoyed very low disposal costs for the residential waste collect in Harrisburg.

The cost of the operation has also been enhanced by [illegible] use of on site ash disposal. But there is very little room left [illegible] the city's lined ashfill and, with the plans for a retrofit of the old facility that will result in a renovated plant ready for another 20 years of life almost ready to be implemented, the city must identify a means of ash disposal.

Mayor Stephen Reed has always been committed to engendering alternate energy sources (the city now recovers methane from [illegible] water sludge digesters and makes electricity, and the city is currently developing a 25 megawatt hydroelectric project on the Susquehanna River). He understands that the region will benefit in many ways by the continued operation of the waste-to energy plant. However, issues of environmental protection and the cost of ash management have made the retrofit project's economics problematic. As a result, I was authorized to fund a study with the intent of reviewing the many ash treatment approaches and their possible application to a usable end product.

We retained a consulting engineer and also advertised in all of the most widely circulated trade publications asking for those with ash fixing processes to respond. We received numerous responses. An evaluation of the various methods was made and two of the most promising chemical fixation techniques and two of the most promising vitrification process were selected. The processes were subjected to a battery of tests for both the degree of stabilization of the "bad actors" and the potential for reuse in a variety of applications such as asphalt, concrete, brickmaking, road barricades, and marine structures.

THE PROBLEM

There are two known methods of municipal solid waste (MSW) disposal: combustion for volume reduction and energy recovery (waste-to-energy; WTE), or landfill of raw MSW. Many nations and some states in this country have found that waste-to-energy is vital to their solution of the disposal problem both because of a reduced dependance on landfill space and as supplement to the energy requirements of the area. The combustion of MSW and the associated gaseous emissions controls, however, produce a grey to black material commonly referred to as ash or residue. And, the ash being greatly increased in density compared to the original MSW, sometimes contains concentrations of lead and cadmium that exceed the regulatory requirements that differential between a non-hazardous and a hazardous waste.

Landfilling of raw MSW is not without its drawbacks as well. The problems associated with the landfilling of raw MSW are real and serious; problems with both leachate and airborne emissions of various organic substances have, in numerous locations, caused a pronounced degradation in the local environment. Comparing data from seven sanitary landfills and from four ash landfills indicates that leachate from ash fills contains lower concentrations of salt and metals than leachate from MSW landfills.[1]

There is, however, a well organized group of environmentally conscious people that have focused on the WTE technologies with a strong opposition to that solution. Oddly, these groups do not rationally evaluate the data associated with both of the MSW disposal solutions, but, in fact, have ignored the negative aspects of landfilling MSW and have focused on what they perceive to be the far more serious environmental problem; MSW combustor ash.

The result of such a well organized program is that MSW ash is considered a waste that must be landfilled in a far more complex and expensive landfill than raw MSW.

As a result, the cost of operating a modern well run waste-to-energy facility is strongly influenced by the cost of ash disposal. Even with the volume reduction of some 85% to 95%, the cost of ash disposal (not including transportation costs) will add between $15 to $40 onto the tipping fee. With all of the other costs associated with the construction and operation of a modern waste-to-energy plant, the high cost of ash disposal has driven the tipping fees required to make expenses up into the $60 to $100 ranges.

Landfills, even with the more stringent regulations regarding the protection of ground water and the control of air borne releases of organic compounds and underground gas migration, can cover the costs of construction, operation, closure, and post closure maintenance with tipping fees of between $25 to $60.

With the exception of certain regions of the US that either contain extremely dense populations or have extraordinarily vulnerable ground water and therefore cannot find a place to site a MSW landfill, the future of WTE is very much in doubt.

The solution to the problem of re-gaining a competitive tipping fee is the elimination of the need to landfill the ash. A true zero discharge system that completely recycles all of the useable material in the incoming waste, all of the liquids produced in the waste-to-energy process, and all of the ash left after combustion and flue gas cleaning is the solution. The first two items in the previous sentence have been achieved; it is the recycling of the ash that is the problem.

Proponents of recycling suggest that all of the material in the MSW waste stream can be recycled. In fact, this is what is required in order for us to enter the 21st century and provide this planet with responsible management of MSW (and conservation of natural resources). Total recycling can be achieved through ash reuse.

However, the same concerns for toxic constituents that have caused the landfilling of ash to become expensive have prevented any real progress in this country in developing a reuse technology. Even if we achieve a reasonable success in removing the obvious bad actor in the MSW (batteries are responsible for 70% of the million tons per year of lead consumed, and for 29% of the cadmium used in the US) we will still have to deal with other less easily removed sources: picture tubes from TV's (30,000 tons per year of lead), solder for electrical parts (4,333 tons per year of lead), plastics (PVC wire insulation), highway yellow and white paint (14,000 tons per year of lead), demolition debris from old homes with lead based paint, lightbulbs, automobile wheel weights, wine bottle closures,[2] and plastics (26% of cadmium in MSW is from plastic[3]).

Even if the metals are removed or reduced, there are concerns regarding PCDD's and other possible toxic compounds that are resident in the ash[4]. In a "flash report" memorandum dated October 5, 1987 (from the "Inspector General", USEPA to Mr. James Barnes, Deputy Administrator) regarding a ship load of ash from Philadelphia, there is the statement "There is a reasonable potential for the ash to cause significant damage to the wetlands environment and human health...analysis of the dioxin content of the ash...show the ash to be considerable more toxic than previously publicized." The material on the ship was to be used to build roads but, partially due to EPA intervention, the material never officially left the ship. The EPA memo goes on to say "the threat will be the long term exposure of the road base itself...in the long run the ash will still be in substantial contact with the wetlands environment." Interestingly, in Denmark, testing has shown that storm water run-off from the surface of roads was more contaminated that the water that passed through the ash used as the subbase under the pavement.[5] Exposure of ash to fish and rats has resulted in classifying the material an "extremely hazardous waste"[6].

Considering both the ash regulatory climate and the strong public pressure regarding the perception of ash as highly toxic material, permanent isolation of the "bad actors" from the environment must be achieved prior to the creation of any reusable product.

METHODS OF IMMOBILIZING THE TOXIC CONSTITUENTS

CHEMICAL METHODS

Two methods (both proprietary) were selected for evaluation.

Method (A) resulted in the recent delisting of electric arc furnace dust (EAF) . EAF has very similar characteristic to MSW flyash both in types and concentrations of heavy metals. Since the process has been successful in removing a listed hazardous waste from 40 CFR 261.31 and 261.32, we felt that the process merited review by our engineers and chemists. The fixed material is a chocolate brown color and is hard with a break strength of about 280 psi.

Method (B) is a process developed specifically for use with WTE flyash. The process claims complete and irreversible immobilization of the heavy metals. Since the process is being promoted by a firm reputable in the management of various chemical and hazardous wastes, we felt that the process merited review by our engineers and chemists. The fixed material is a grey color and has a sandy consistency.

Extraction Procedure Toxicity (EP TOX) tests on Harrisburg flyash and mixed ash after the application of methods (A) and (B) are below.

(values in mg/l)

	Treated (A) Fly Ash	Treated (A) Mixed Ash	Untreated Mixed
Ag	.037	.029	.041
Cd	.043	.031	.62
Cr	.27	.23	.29
Cr^{+6}	.07	<.01	<.01
Cv	.08	.18	5.0
Pb	<.1	<.1	7.8
Ni	<.05	<.05	.32
Zn	.028	.054	195.0
As	.18	.08	.07
Fe	.20	.32	18.2

(values in mg/l)

	Treated (B) Fly Ash	Treated (B) Mixed Ash	Untreated Mixed
Ag	.043	.023	.041
Cd	.044	.035	.62
Cr	.34	.26	.29
Cr^{+6}	.06	<.01	<.01
Cv	.12	.21	5.0
Pb	<.1	<.1	7.8
Ni	<.05	<.05	.32
Zn	.018	.043	195.0
As	.13	.08	.07
Fe	.37	.33	18.2

It was noted that the initial pH of the method (A) treated material was 12.3 and, after all of the acid was consumed by the EP TOX test, the final pH of the treated material was 9.3. The initial pH of the method

(B) treated material was even higher, 12.6. 12.5 is considered corrosive.

It has been noted in previous research that the lead and cadmium in the ash will bind in the presence of chlorides and sulfates and will render them immobile as long as alkaline conditions are maintained[8]. Figure (1) also shows that an extremely high pH will also allow mobility of the metals.

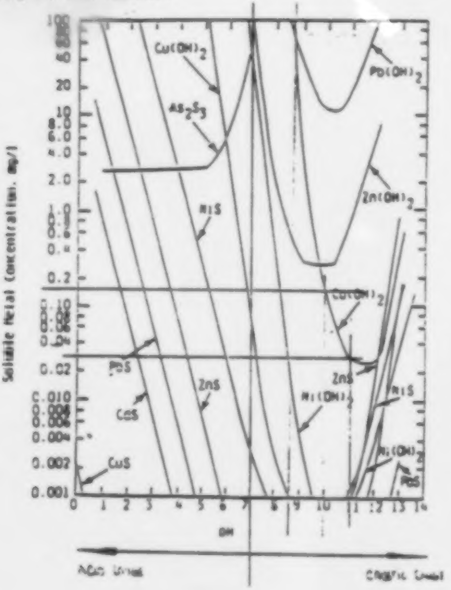

Figure (1)

The methods (A) and (B) treated flyash was then forced to a pH of 5.0 and maintained for the 24 hour leaching period by the use of 3N HCl to supplement the acetic acid and minimize volume change. The material showed substantial leaching toxicity after the excess alkali was neutralized.

Modified EP TOX Test - pH forced to 5
values in mg/l

	Method (A)	Method (B)	EP TOX Limit
Arsenic	.61	.72	5.0
Barium	<.5	<.5	100.
Cadmium	8.7	6.2	1.0
Chromium	.11	.32	5.0
Lead	23.	32.	5.0
Mercury	<0.001	<0.001	0.2
Selenium	.12	.14	1.0
Silver	.19	.20	5.0

The fact that the metals leached out when the material was maintained at a pH of 5.0 for 24 hours caused us to question the existence of any real chemical fixation in the processes. Several EP TOX runs were made on the mixed ash after 10% by weight of $CaCO_3$ (calcium carbonate) was mixed with the ash. Previously, the sample failed with a value of 7.8 mg/l lead and after the 10% mix the lead was <.01 mg/l and the sample cleanly passed on all parameters with metal reductions of 3 to 100 times. Use of a stronger base (CaO) may have resulted in even lower values.

From this limited testing, it would appear the two of the most promising stabilization methods are very much pH dependent. The existence of any chemical fixation is questionable. Both of the treated samples [method (A) and (B)] had far more than enough basic material to neutralize the acid in the EP TOX test.

Pulverization of the stabilized products did not change the results of the EP TOX testing.

Limited funds prevented the testing of the fixed material for dioxins. However, nothing in either process suggests that any change in dioxin values would occur.

Capital costs for the equipment required for the mixing are similar to a concrete batch plant. A 150 ton per day operation would cost about $2.5 million for equipment and installation. Operation costs, according to the method (A) inventors is about $23/ton. Method (B) would not divulge the operational costs.

MELTING AND SOLIDIFICATION

Two methods (both proprietary) were selected for evaluation.

Method (X) uses a continuous electric furnace. The process utilizes a traditional continuous feed furnace with three electrodes. The emissions from the process are minimized because the cold material is introduced in the top of the crucible and act to condense the vaporized metals that leave the molten surface. Power consumption equals about .23 KWH/lb. or 460 KWH/ton. The end product is a black obsidian like material that is hard and shiny. EP TOX testing resulted in no detectable metals. The final product also resulted in an increase in density of from 52 pounds per cubic foot to 163 pounds per cubic foot. The material is very hard (6.5 Mohs scale)

Method (Y) uses a continuous feed furnace heated with oil or gas burners. The furnace has been designed specifically for the melting of MSW ash. Emissions from the process are directed into the combustion air ducts for the waste-to-energy facility and the particulate is recycled through the furnace. Energy requirements for the process are 35.2 gallons of #2 oil per ton. The end product is identical to method (X). Also, EP TOX testing resulted in no metals leaching.. The density increase for method (Y) is very similar to method (X); from 52 PCF to 172 PCF.

Again, limited funds precluded any testing for dioxins. However, the ability of glass to absorb metals and other compounds into its matrix[10] combined with high temperature of the process (2400-3000°F.) would suggest that dioxins would either be destroyed or encapsulated within the glass matrix.

Capital cost for a method (X) is $3,940,000 for the equipment and $2,500,000 for the installation. Operating cost for method (X) is $38/ton (that includes power and personnel).

Capital cost for method (Y) is estimated to be similar to method (X) with the exception of the need for high voltage/high amperage connections. Operating cost based on oil are $40/ton (fuel and personnel); based on natural gas the cost is $35/ton.

REUSE OF THE STABILIZED MATERIAL

Each of the applications evaluated can make use of the ash in the final form after the various stabilization methods methods have been applied. Fine tuning of methods (A) would most probably result in much higher break strengths[11]. Method (B) may or may not be applicable to reuse because of the sandy nature of the final product. Methods (X) and (Y) can either be molded directly from the melt or can be quenched when hot to produce a "frit." Frit was produced from the ash we had processed and was evaluated as a possible aggregate.

Although it would appear that method (A) does not satisfy the concerns of the regulatory agencies and the public in regards to degree of isolation of the metals and dioxins from the environment, the fact that it is a hard material caused us to include the method (A) product in the evaluations.

The end products evaluated are paving base, concrete, and brickmaking.

PAVING AND CONCRETE

Research done by the Federal Highway Administration in the 1970's resulted in the construction of several streets in Harrisburg using ash from the Harrisburg waste-to-energy plant[12]. A 180 foot long strip of bituminous sub-course containing ash was laid as part of a resurfacing project in 1973. The same year a complete block of a residential street wearing surface in south Harrisburg was laid with a blend of natural aggregate and ash. Both streets are still used, and with the exception of the normal potholes associated with freeze thaw cycles, the streets are in good condition. The Harrisburg Director of Public Works inspected both locations and found no obvious problems. Furthermore, the plant growth along the streets seems normal. Constituent analysis for heavy metals in the plant growth by the roads in question compared with the same plants elsewhere showed no difference. Obviously, the material that was used in the early 1970's was not stabilized to immobilize the metals. The methods (A), (X), and (Y) all produce an aggregate that would be suitable[13,14].

Small amounts of bitumenous mix was made using the each of the three methods. The mix was was made at 5.5, 6.0, 6.5, and 7.05% asphalt.

Method (A)

Bitumen (%)	5.5	6.0	6.5	7.0
Marshall Stability (lbs.)	735	875	1215	890
Flow (1/100 in.)	13	17	16.7	17.9
Wt. per Cu. Ft. (lbs)	132.2	135.2	137.0	135.0
Air Voids (%)	11.6	8.7	7.1	7.0

Methods (X) and (Y)

Bitumen (%)	5.5	6.0	6.5	7.0
Marshall Stability (lbs.)	690	800	1012	780
Flow (1/100 in.)	13.1	16.2	16.3	17.9
Wt. per Cu. Ft. (lbs)	134.5	138.9	140.0	139.2
Air Voids (%)	10.6	7.7	6.3	6.7

All test were successful as measured by Marshall requirements.

The suitability of the stabilized products for concrete aggregate (not as bitumenous concrete aggregate as is discussed in the above paragraph) was to be a part of our research. Unfortunately, time and funding constraints prevented that portion of our plan from being implemented. Based on research that we have reviewed[15], the material is suitable for a variety of end uses. Marine tetrapods and artificial reefs are currently being tested near Long Island. The break strengths of the concrete would suggest that highway barricade ("Jersey Barricades") would also be a useful end product.

BRICKMAKING

The City of Harrisburg did undertake creation and testing of bricks made with pulverized method (A) product and the fine frit of methods X) and (Y) in varying percentages. The stabilized products were mixed with w/w standard hydrated lime and the water content of the resultant mixture adjusted to 4.0%. Miniature specimen bricks were then prepared at normal commercial press loading.

The bricks were then subjected to standard autoclave curing.

The cured specimen were tested for a dry compression strength [16]

The data for the testing of the strength of the bricks has not been received at the time of the writing of this paper. The data will be presented at the conference.

CONCLUSION

The solution to the concerns regarding the material contained in MSW combustor ash is easy. The problem is doing it in a way that is cost effective.

Obviously, if the material is to be placed at a landfill designed to meet current ash regulations, it makes no sense to spend more money stabilizing the material. However, if the ash is finally characterized as a hazardous waste, the logistics many change depending on the cost of disposal.

In 1992 it is projected that the US will produce 14,600,000 tons per year of ash from waste-to-energy facilities[18]. In 1990, the annual demand for aggregate in the US will be 1,500,000,000 tons[17]. The amount of ash available compared to urban aggregate demand is not large enough to present a serious business conflict to the fixed investment of the vertically integrated highway construction and materials industry. The current cost of natural aggregate is about $5.00/ton. The cost of a stabilized material made from MSW ash ranges between $40 and $65/ton.

Although the quality of the end products is adequate, the cost of production and the cost of amortization of the equipment cannot compete with naturally produced products.

The bottom line under today's economy of ash management suggests that the end product must be worth at least $65/ton.

Bricks are currently valued at .$24 each or about $96/ton. Further research on the suitability of making brick from stabilized MSW ash is highly recommended. If the percentage of ash can be high enough, the process may be cost effective.

Finally, the dramatic reduction in volume achieved by the melting and solidification of the ash (from 3 to 5 times less volume) suggests that even under a landfilling scenario, the ability to put 3 to 5 times more ash in the same space may offset the cost of installing and operating the melting furnaces.

The City intends to further investigate the economics of end use products made from MSW combustor ash. The solution to the ash disposal cost problem must be found or waste-to-energy will fade away as large landfills are sited and built and railhaul methods for moving MSW are perfected.

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MUNICIPAL WASTE COMBUSTION ASH: TESTING METHODS, CONSTITUENTS AND POTENTIAL USES

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INTRODUCTION

This paper focuses on the content of combined ash, which represents the non-combustible portion of municipal solid waste. The term "ash" refers to all the solid residue streams of a municipal solid waste combustion system. This included: the bottom ash (approximately 85 to 95 percent of the total), also referred to as "clinker"; ash that collects on boiler, superheater and economizer tubes often referred to as "soot"; ash that is removed from the flue gas via the boiler passes, and the fly ash collected by the air pollution control equipment. The last type of ash is a combination of the solid material, as well as the reacted and unspent reagent from the acid gas scrubbers, that is collected after reaction in the scrubbers by the particulate collection equipment (e.g., the fabric filter or electrostatic precipitator). Bottom ash mainly comprises bottles, cans, auto parts, broken appliances and a myriad of other things that do not lend themselves to complete combustion. Fly ash originates from the ever-present fraction of inorganic compounds in paper, wood, plastic, rubber and food wastes. For example, there are clays in papers, stabilizers in plastics, pigments in printing, inks and mineral salts in vegetables and other products. In other words, the constituents that comprise the two types of ash represent the unavoidable, inorganic, by-products of everything that is thrown away.

By volume, the ash is approximately one-tenth to one-twentieth of the original incoming municipal solid waste. By weight, it represents approximately one-fourth of the original amount. It is much denser than raw solid waste and takes up much less space for ultimate disposal.

ENVIRONMENTAL BENEFITS OF COMBUSTING SOLID WASTE

Volume reduction, converting large amounts of solid waste material into smaller volumes, is a basic environmental protection practice that has been used for thousands of years. Many federal solid waste regulations are based on this premise and it has been a preferred solid waste management option for more than a century. Unfortunately, despite successful reduction and recycling programs, the United States will still need to combust the remainder of the waste stream.

Combusting municipal solid waste and converting it into energy has three basic purposes. First and foremost, the high temperatures accomplish sterilization and deny food and habitat to disease vectors such as rats and flies. Secondly, volume reduction is successfully accomplished. And, thirdly, useful energy is captured. Throughout history, municipal solid waste was combusted to control disease and reduce volume. Up until the 1960s, open dumps were often set afire as a method of disease (vector) control and to make room for more garbage. Adverse repercussions such as brush fires, rat migration, air pollution, and worker fatalities led to public health problems that made open burning unsafe. However, burning municipal solid waste in a carefully-controlled furnace and managing ash efficiently can satisfy the original objectives without causing environmental harm. While volume reduction to reduce the burden of final waste disposal is essential, combustion can also convert municipal solid waste into energy that can be recovered.

The conversion of raw, unprocessed municipal solid waste into a smaller volume of ash results in long-term land preservation because ash disposal requires less space. There are many differences between disposing of raw municipal solid waste and landfilling ash.

Many environmental problems associated with municipal solid waste disposal can be reduced, mitigated, or eliminated with combustion. Landfilling of raw municipal solid waste generates odors, methane gas, and many other toxic and reactive gases. All these gases originate from the biological decomposition of organic matter. While odors are more annoying than dangerous, methane gas migration from poorly-designed and/or operated landfills has caused explosions in structures located nearby, resulting in millions of dollars of property damage and loss of life. By contrast, waste-to-energy ash is biologically inactive, generating no odors or explosive, toxic, or reactive gases from the landfill.

Potential ground water contamination can occur from any material disposed of in a landfill. This contamination is the result of moisture contained in the material, or rain, or surface water, infiltrating the material and leaching out toxic contaminants. Because organics predominate raw municipal waste, leachate from a landfill is also highly organic. In addition, it is acidic because of the biological activity within the landfill. The organic acids have a much greater potential to carry toxic compounds and elements into groundwater than the leachate from an ash landfill. In contrast, ash leachate that forms in small amounts (since infiltration through dense ash is very slight) contains only inorganic compounds -- salts and metals -- that do not easily travel through soil. By locating the landfill in the proper hydrogeological setting and/or by using liners and leachate collection systems, both leachates can be prevented from entering groundwater. Because ash leachate is predominantly inorganic, it can be contained with a simple lining system, and is not difficult to treat. Subsequently, the potential for groundwater contamination is much less than from a raw municipal solid waste landfill.

This has already been demonstrated in other parts of the world: Wurzburg, Germany; Marion County, Oregon; Denmark and Sweden. In addition, the United States Environmental Protection Agency (EPA) is investigating ash leachate in a joint study with Marion County at the county's Woodburn monofill.

Dust represents another problem at landfills that can be reduced with combustion. Municipal solid waste is a very dusty material. When it is deposited in a landfill, without adequate controls, bits and pieces blow around. The compaction process requires driving large bulldozer-type vehicles over the municipal solid waste until it is compacted, which may increase dust generation. On the other hand, the semi-wet ash is delivered to the landfill and does not result in any fugitive dust. Since wet ash is so dense, it does no require much compaction. Consequently, very little airborne material is released into the environment during ash disposal.

Unlike ash residue, raw waste undergoes biological decomposition during the many years it remains placed in the landfill, volume reduction occurs gradually over time; it is an ongoing, slowly-evolving process. This continuous shrinkage causes the surface of the landfill to subside and requires extensive reworking of the soil. The subsidence also results in increased infiltration of rain water, which in turn produces more leachate. An ash landfill, however, becomes a stable mass within days of initial placement and requires only simple maintenance of the final cover. No subsidence whatever occurs. Modern waste-to-energy facilities reduce the volume of material and mitigate the potential public health problems associated with the ongoing land disposal of municipal solid waste.

PHYSICAL AND CHEMICAL PROPERTIES OF ASH

Unprocessed municipal solid waste contains varying percentages of inert materials that eventually become the ash or solid residues upon combustion in a resource recovery facility.

The volume reduction achieved by combustion increases the concentration of the metals in the ash versus the unburned municipal solid waste. The total amount of metals, however, have not increased. The increase in concentration cannot be any higher than the weight reduction (that is a 4-to-1 weight reduction of waste to ash, or 80 percent, increases concentration by a factor of five). Regardless of whether it is municipal solid waste that is landfilled, or ash from a resource recovery facility, the total amount of metals going in will remain the same. No data is available to suggest that the combustion process changes the properties of the metals to make them any more dangerous. On the contrary, leachate data from raw solid waste landfills suggests higher metal mobility because of biological activity and the presence of organic acids. From the available data on ash collected by Ogden Martin, the following list shows average metal concentrations in categories of major, minor and trace constituents:

METALLIC ASH CONSTITUENTS

MAJOR	%	MINOR	%	TRACE	%
Aluminum	3	Copper	0.1	Arsenic	0.003
Calcium	8	Lead	0.2	Barium	0.05
Iron	10	Manganese	0.6	Cadmium	0.003
Sodium	6	Molybdenum	0.1	Chromium	0.02
Silica	30	Potassium	0.4	Mercury	0.0006
		Titanium	0.7	Selenium	0.004
		Zinc	0.3	Silver	0.0006

In addition to the metals listed above, ash contains many other elemental compounds that make up the bulk of the material. Oxygen, sulfur, chlorine, which occur as oxides and sulfates, and chlorides account for more than 30 percent of the ash. There are also trace elements that can be detected at the lowest detection levels of modern analytical chemistry. This is also true of organic compounds, other than unburned or fixed carbon.

Based on an average of available data, the physical properties of ash can be tabulated as follows:

Density	one or two tons per cubic yard
Specific gravity	approximately one to three
Moisture	15% to 25%
Grain Size	10% similar to clay
	40% similar to sand
	30% similar to gravel
	20% larger than gravel
Permeability	10^{-3} to 10^{-4} cm/sec as landfilled
Texture	Wet concrete-like or wet sand and gravel-like

ASH MANAGEMENT AT AN OPERATING FACILITY

The Martin GmbH mass-burn design encompasses the closed-system concept to handle the process residue streams so that no employees, visitors or nearby residents will be exposed to airborne dust. All hoppers from the boiler to the air pollution control device have sealed, air-locked valves that transfer the fly ashes and/or dry scrubber reagent and reaction products to sealed screw or drag conveyors. In turn, these conveyors deliver all of the combined fly/bottom ash stream to the sealed discharger at the lower end of the furnace; just below the level where the bottom ash drops off the stoker grate.

As the bottom ash falls down the ash discharger chute into the quench water bath, the fly ash is captured, moistened and combined with the bottom ash before it is hydraulically discharged by a ram. The ram extrudes and dewaters the ash prior to dropping it on the conveyor system. At this point, the ash has the consistency of wet concrete.

As it moves outside the boiler building, an enclosed conveyor transfers it to the ash storage building or another area to await transfer to covered, water-tight trucks or containers for eventual utilization or landfill burial. At any point, where the employees or the public may be exposed, the ash system is either sealed or the ash moistened and enclosed in a building or conveyor.

ASH CHARACTERIZATION

Since the state environmental regulatory agencies have difficulty accurately predicting the potential adverse effects of burying waste, these governmental agencies typically have a two-pronged approach to the problem. Specific types of wastes, with high potential for causing environmental harm, are listed as "hazardous".

Secondly, regarding other wastes including ash, these agencies have established four broad characteristics to use in identifying waste that must be managed as "hazardous waste". The characteristics include ignitability, corrosivity, reactivity, and extraction procedure toxicity. This last characteristic is often applied to municipal waste combustion ash. The EP toxicity characteristic is determined by a devised laboratory test that attempts to mimic the landfill environment and disposal scenario of five percent unknown waste and 95 percent raw municipal waste in an unlined landfill.

Many problems occur with tests that take a small sample of waste, subject it to a laboratory analysis and then use the results to predict how larger quantities of hypothetical waste will really behave. For example:

- How do you obtain representative samples?
- How so you prepare these samples for laboratory analysis?
- What type of test is most appropriate for the manner in which the waste will be managed?
- How should varying results be interpreted?

Researchers in the waste management field have proposed additional tests that may be more appropriate. Currently, however, the ultimate test is one of sampling the actual leachate/groundwater from the managed (or mismanaged) land disposal unit.

The EP Toxicity test is the current test method employed by the EPA to determine if an unknown waste that may be subject to leaching in a landfill should be managed as a "hazardous waste". The Toxic Characteristics Leaching Procedure (TCLP) and other tests have also been proposed (published in the Federal Register and implemented for certain regulations as the land ban for certain hazardous wastes). Indeed, after much consideration, some agencies have eliminated or exempted waste-to-energy ash from laboratory testing.

Before considering what test should be used for regulatory or research purposes, a method must be available to obtain a representative sample. One cannot obtain an accurate sample by simply "grabbing" a small portion of the ash residue from a resource recovery facility, test it and declare that it is "hazardous". There is a very deliberate and detailed sampling procedure mandated by Section 1 of SW-846 (Test Methods for Evaluating Solid Waste: Physical/Chemical Methods, U.S. EPA, Publication #SW 846, July 1982 as amended). Only by following these guidelines can one attempt to obtain representative ash samples and conclude within a certain degree of confidence, that the ash does or does not exhibit the characteristic of EP Toxicity, that it is hazardous by the TCLP, or by any other leaching procedure.

Even when Section 1 is closely followed and an extensive sampling program is performed, it may not necessarily provide representative samples. To adequately state the results within a 90 percent confidence limit, the number of samples cannot be determined until all the results of the samples are analyzed statistically. After following the sampling guidelines in SW-846, a difficult task of sample preparation must be applied.

The preparation methods in SW-846 result in the collection of many samples each weighing 50 to 100 pounds. These must be reduced in size to pass through a nine millimeter sieve. Typically, this step has been omitted in most ash sampling programs. The samples are usually screened and the subset of the sample, which passes through the nine millimeter screen, is the portion subjected to laboratory analysis. This, however, results in an unrepresentative sample in violation of the procedures in SW-846. Furthermore, some laboratories have failed to maintain the sample's moisture content or have completely dried the samples to ease preparation.

Once a representative sample is obtained, there are many laboratory tests that can be performed to collect data about it. This data is then used to determine leaching characteristics. Different official procedures require different tests for various characteristics that the ash exhibits. These tests include:

1. Percent Moisture
2. Particle Size and Structural Integrity
3. pH
4. Selected Anions (negatively-charged ions) and Cations (positively-charged ions)
5. Total Metals
6. Organic and Carbonaceous Material
7. Leaching or Extraction Tests, such as the EP Toxicity Test, the TCLP Test, a De-Ionized Water Leaching Test, a Solid Waste Leaching Procedure Test or Column or Lysimeter Tests

The extraction fluids are analyzed by atomic absorption or inductively-coupled argon plasma (ICAP). The analytical data are then examined for quality assurance/quality control and treated statistically to determine the results and confidence limits.

As rigorously as possible, OMS has followed the procedures in Volume 40 C.F.R. 261.24 and SW-846 in sampling and analyzing the ash from its operating facilities located in: Tulsa, Oklahoma; Marion County, Oregon; Hillsborough County, Florida; Bristol, Connecticut and Alexandria, Virginia. The ash did not exhibit the characteristic of hazardous waste in any instance.

The TCLP which is under development appears to be more consistent in yielding results. Sample collection and preparation problems are the same as with the EP (grinding portions of the ash that will not break down in the landfill). Consequently, like the EP, the results never represent behavior in the real environment because acid is poured through the ash, rather than dripped through it, over an extended period of time. Although the consistency of the data resulting from leachate procedures in the EP toxicity test have been improved with the TCLP, neither test produces data that simplifies what actually leaches from ash. A more accurate procedure for determining leaching characteristics would involve testing the actual leachate from an ash fill, especially a monofill that contains only waste-to-energy ash residue.

Test data for ash residue from OMS plants has been submitted to the EPA and state regulatory agencies. Lead and cadmium represent the only metals that occasionally show levels higher than the regulatory threshold in some limited samples. To simplify the following discussion, only lead will be addressed. When presenting the EP Toxicity or TCLP data, the value that is important is the upper limit of the confidence level. It is this value that is compared to the regulatory threshold and not individual sample values, even if they exceed the regulatory threshold. The most recent testing of the ash at Tulsa shows the following results for the upper band of the confidence limit on three separate analyses for EP toxicity and are for TCLP:

	EP Toxicity	TCLP	Regulatory Threshold
Lead (mg/l)	2.8	3.0	5.0

for the upper bound of the confidence limit. Of course, this means the average values were actually less.

For Marion County, a comprehensive testing program was performed in November 1986 during the facility's energy and capacity tests. The Marion County data is particularly significant because this facility is the first in the United States using a dry scrubber and high efficiency particulate control equipment (fabric filter baghouse) on a resource recovery facility. The data from Marion County's ash yielded the following results for the upper bound of the confidence limit on three separate analyses for EP toxicity and one for TCLP:

	EP Toxicity	TCLP	Regulatory Threshold
Lead (mg/l)	3.4 to 4.9	0.9	5.0

The tests conducted on the Bristol, Connecticut, facility, which has the same air pollution control configuration as the Marion plant, yielded similar test results. The upper bound of the confidence limit for lead was 2.5 mg/l.

OMS has performed, or has cooperatively studied with regulatory agencies, other tests on the ash from its facilities. With regard to the EP Toxicity organic substances, the levels are below the detection limit. Portions of the ash (i.e., the fly ash) were analyzed by the EPA and determined to contain among the lowest levels of dioxin ever found in fly ash. These extremely low levels correspond to the very low levels of dioxin emissions achieved from the Martin GmbH stoker combustion system without any back-end pollution control, as well as from the stack gases with pollution control.

Limited leachate data is available from existing ashfills and sanitary landfills. The lead levels are lower for ash monofills, but results from both are far below the regulatory threshold of 5 mg/l. The data are as follows:

	Lead (mg/l)
Sanitary Landfill Leachate (Malcolm Pirnie)	0.2 to 1.0
Ashfill Leachate (Malcolm Pirnie)	<0.05 to 0.13
Wurzburg Ash Monofill Leachate	0.002 to 0.05
Sanitary Landfill Leachate (Malcolm Pirnie)	0.2 to 1.0
Ashfill Leachate (Malcolm Pirnie)	<0.05 to 0.13
Wurzburg Ash Monofill Leachate	0.002 to 0.05
*Marion, Ore. Ash Monofill Leachate	0.1 to 0.6
Scandinavian Monofill Leachates	0.001 to 0.1
**EPA Data (Four Sites)	<0.005 to 2.92

*Oregon Department of Environmental Quality
**U.S. Environmental Protection Agency data for Four Anonymous Sites

Measurements for lead and the seven other heavy metals in leachate from a landfill, in which both ash and raw solid waste have been placed (co-disposal), are well below the regulatory threshold(s), and have remained low over time. These lead levels were measured for the 1983-1987 testing of this co-disposal landfill leachate (in mg/10.10, 0.19, 0.06, 0.33 and 0.12). When ash is buried with unprocessed municipal solid waste, the collected leachate still has a very low lead level.

More importantly, data recently collected by the U.S. EPA indicate that proper monofilling of ash has a negligible impact on the environment. The EPA has initiated a long-term study at the Marion County, Oregon, ash monofill. The first year's data show the leachate contained from 0.011 to 0.025 mg/l of lead. To determine if fugitive ash is a problem, soil samples were taken from around the monofill. The lead content of the soil both upwind and downwind of the facility contained 0.01 ppm of lead. Fifty-three (53) ppm lead was detected in the soil along Interstate 5, a few hundred meters east of the site. Background sample of soil east of the Interstate and far west of the facility showed levels of lead in the range of 0.014 to 0.017 ppm.

The EPA also tested for dioxins and furans in the soil around the monofill. The results were similar to tests conducted for lead. No differences between upwind, downwind, and background. The highest levels detected were along Interstate 5.

THE REGULATORY STATUS OF ASH

Because of the changing definition of "hazardous" waste, the regulatory status of municipal solid waste combustion ash has been subject to debate over the past ten years. Prior to the Resource Conservation and Recovery Act of 1976 (RCRA), ash was primarily regulated by individual states or local jurisdictions as municipal solid waste. It was often approved as cover material for municipal solid waste sanitary landfills.

With the implementation of RCRA and the development of the federal hazardous waste management program, waste classification took on a new meaning. Congress and EPA split up the universe of waste into two categories. Those wastes, subject to management under Subtitle C of RCRA (hazardous wastes), and wastes subject to management under Subtitle D (non-hazardous wastes). Household waste and the ash residue from processing household waste were specifically excluded from Subtitle C.

This meant that household waste and incinerator ash was non-hazardous by definition. Since municipal solid waste, designated for a resource recovery facility, represents a combination of household waste and non-hazardous commercial and industrial waste, exclusion under the federal hazardous waste rules (Subtitle C) was thought to be unnecessary and therefore unavailable. Consequently, it became a requirement for the owners/operators of municipal solid waste combustion facilities to determine if their ash exhibited a hazardous waste characteristic by this difficult and onerous testing method.

When Congress enacted the Hazardous and Solid Waste Amendments of 1984, this regulatory difference between household waste and municipal solid waste ash was clarified. In this law, the ambiguous definition of "household waste" was clarified. The law states that, if a facility takes only household and non-hazardous commercial and industrial waste and has a program to prevent Subtitle C hazardous waste from being accepted, it is not deemed to be generating, treating or otherwise managing hazardous waste. In other words, the ash would not be subject to management as a hazardous waste, and testing to determine its regulatory status would not be required.

When EPA placed the law into the Code of Federal Regulations (CFR), they did not interpret it in the same way as Congress intended. EPA stated that, if the ash was tested and exhibited the characteristic of a hazardous waste, it had to be managed as such. This position has caused a great deal of confusion. If it exhibits a characteristic of hazardous waste, is the ash exempt from requirements to be managed as hazardous since it represents the by-product of municipal household waste? Or, is it to be managed as a solid waste in all cases? EPA is presently re-evaluating its decision, developing a new series of characterization tests to determine the regulatory status of ash, and developing environmentally-sound design criteria for ash burial.

Numerous states have taken similar positions. New York, Massachusetts, Tennessee and others have stated publicly that they believe the congressional clarification applies to the combined ash. The state of Oregon, well-known for some of the toughest environmental standards in the United States, believes the present testing requirements in 40 CFR 261 (which stipulates performance of the EP Toxicity test and other leaching procedures for hazardous wastes) are invalid for determining ash characteristics and should not be used to determine regulatory status. Other states are taking the same position. All state and federal regulatory agencies that are addressing this subject believe that efficient ash management is essential regardless of the outcome of various characterization tests.

There is no environmental exclusion from testing ash, but what is considered "efficient management" varies across the board nationwide. Consequently, regulators are developing proper ash management criteria for compliance. These criteria focus on prevention of groundwater contamination through the use of lined landfills. While monofilling ash is the most desirable option, it is probably an overly stringent requirement considering that it is unlikely that co-disposal leachate would penetrate a well-designed system, particularly one which incorporates a leachate collecting system as well as the required linings.

Both Congress and the EPA are in the process of further clarifying the regulatory status of ash and developing management standards. In numerous proposed bills, Congress has clearly stated that ash should be managed as a Subtitle D (non-hazardous) waste. In addition, all the bills require some type of lining, leachate collection, and groundwater monitoring systems for ash disposal. It is doubtful that these bills will be voted into law this session of Congress, but legislation as important as this should move in the spring of 1989.

EPA has released draft guidance on the handling, transport, storage, and disposal of ash. This guidance includes recommendations that ash containers and transport vehicles be leakproof and provided with tight coverings; that groundwater monitoring be performed at all ash disposal facilities. These liners and disposal recommendations are as follows:

- For fly ash disposed separately, disposal should be at a monofill with a double liner system.
- For combined ash or bottom ash, disposed of in a monofill, either a composite liner or a clay liner with special environmental or operating features should be used.
- For combined ash or bottom ash codisposed with garbage, a double liner or a composite liner, with pre-disposal ash treatment or source separation to reduce metals content prior to combustion is the preferred method.

ASH UTILIZATION

As discussed previously, ash contains many valuable metals and the non-metallic fraction has properties very similar to sand and gravel. These characteristics lend themselves to potential economic benefits. Ferrous and non-ferrous metal recovery using magnets, screens and other mechanical processes is used at many municipal solid waste combustion facilities worldwide. The techniques for recovery of the larger metallic components (e.g., greater than one inch) are well developed. Metals are not recovered on an industry-wide scale in the United States because of depressed scrap metal markets (i.e., installation and operation of metal recovery equipment is dependent on local scrap metal markets). OMS investigates the local scrap markets to determine the economic viability of metals recovery from all its facilities as they enter commercial operation.

There are many metals in the ash that could only be recovered through complex and laborious processes. Metals such as cadmium, lead, zinc, copper, silver and gold are recoverable by using chemical techniques similar to those used in the minerals industry. OMS is currently investigating the feasibility of recovering these metals from the ash with various industrial concerns and research laboratories.

The major component in the ash is the inert, non-metallic fraction. Because the properties are similar to traditional aggregates, ash is commonly used as a substitute for conventional aggregate in Europe. In the mid-1970s, the Department of Transportation (DOT) researched the potential for use in the construction and maintenance of highways. Today, Marion County is pursuing the same option. In Europe, bottom ash is often used in asphaltic paving material and combined bottom and fly ash in concrete. Screened bottom ash is also used as road bed and common fill material. Perhaps, the best use of ash is as aggregate in Portland cement concrete. Municipal solid waste combustion ash has excellent properties for use in concrete itself; it is pozzolanic, which is to say it forms a weak cement-like matrix. This inherent property could be of interest to concrete block manufacturers. OMS is working with two universities to develop proper screening techniques and mixture proportions for cement blocks. Leachability of metals from the blocks will be an important issue that will require resolution before they are widely manufactured. In addition, OMS and one university are determining the likelihood of leachability of metals from the completed blocks.

There are many potentials for the use of combined ash. While the technical problems associated with the various utilization scenarios can be resolved, the institutional problems are more difficult to address (i.e., markets for the metals and public acceptance of the ash as aggregate). Lessons can be learned from Europe and Japan and other waste product utilization programs. In Japan, ash residue is used to make artificial reefs and man-made islands. Thus far, metals have not entered the food chain and subsequently pose no significant risk to aquatic life or human beings. While the United States may not necessarily need ash islands or reefs, there are many other potential uses that would fulfill material needs in the construction, manufacturing or chemical industries.

SUMMARY

Ash is the remaining incombustible residue representing five to ten percent by volume and 15-20 percent by weight of the municipal solid waste stream. It is a biologically inert, dense material that can be managed in a more environmentally sound manner than raw solid waste. Present testing methods do not adequately simulate what occurs when ash is placed into a controlled landfill unit. As indicated by leachate data from actual ash fills, potential ground or surface water contamination from well-managed ash disposal units does not appear to be a problem. Before uses of combined ash can reach their fullest potential, public awareness and a better understanding of resource recovery ash characteristics is necessary. This can only be achieved through governmental leadership at state and local levels, where community education is most effective.

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THE PHYSICAL, CHEMICAL AND ENVIRONMENTAL PROPERTIES OF COAL-RDF FLY ASH

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ABSTRACT

Refuse-Derived-Fuel (RDF) consisting mainly of waste paper and plastic is a viable fuel source for the production of power. An experimental test burn substituting coal with RDF was done by Public Service of New Hampshire at the Merrimack Power Station.

Five Percent and ten percent RDF was substituted, on a BTU basis, for coal in the test burns. Twelve (12) test burn days were run with 4 days of 5% RDF and 8 days of 10% RDF. The effect of the coal-RDF fly ash on portland cement concrete as a means of ash stabilization and utilization was investigated.

The chemical and physical properties of the coal-RDF fly ash as related to its use in concrete were found to be comparable with ordinary coal fly ash except for increased cadmium and lead contents. Cadmium and lead content averages were 5.08 ppm and 102.55 ppm for the 5% RDF and 7.76 and 198.28 ppm for the 10% RDF respectively. The pozzolonic activity index of coal-RDF fly ash increased. Compressive strength was found to be less at 28 days and about the same after 15 weeks when compared to conventional portland cement concrete. Leaching of RDF fly ash concrete with a synthetic acid rain did not release a significant amount of lead and cadmium.

INTRODUCTION

Municipal waste disposal is a major problem in most communities throughout the nation. Many communities have invested in incinerators as a means of reducing their waste. The residue from burning municipal waste is normally landfilled but landfill spaces are fast becoming a rarity and new ways of disposal have to be found. The residue from burning municipal waste, fly ash and bottom ash, contain many elements of hazardous and non-hazardous metals and materials such as glass, plastics and solid metals, due to the wide variety of the mixture of the waste burned as well as the design of the incinerator. Burnable municipal wastes, termed Refuse-Derived-Fuel (RDF), have high BTU values and are viable sources of energy.

Fly ash produced while burning coal in a power plant is a good pozzolan for concrete if it meets the specifications of ASTM C-618. Since RDF is expected to contain higher heavy metal content, the fly ash produced from burning coal and RDF may have a significant increase in heavy metals such that it may be an environmental problem. Norton et al, in their studies show that the heavy metals concentration in coal-RDF fly ash increases, but if only a small percentage of RDF is used to replace coal, there is no great environmental concern (1). Substituting a small percentage of RDF for coal in a power plant would therefore be expected to produce fly ash that has chemical and physical properties meeting the specifications of ASTM C-618.

EXPERIMENTAL PROCEDURES

COAL-RDF FLY ASH SAMPLE

In the summer of 1987, Public Service of New Hampshire ran a test burn partially substituting coal with RDF utilizing the full load of their Unit 1 facility at the Merrimack Power Station. The unit uses cyclone firing with a fuel full load of 84000 pounds per hour, and the power output of 120 megawatts. The RDF, termed "fluff", consisted of shredded municpal solid waste with ferrous and noncombustible materials removed. A typical analysis of RDF is shown in Table 1. The coal was pulverized bituminous coal from West Virginia with a typical analysis as also shown in Table 1. Both the RDF and coal analysis were done according to ASTM standards and the metals were analysed using Atomic Absorption Spectrometry. The test burn utilized 5 and 10 percent RDF coal replacement by BTU content with four days at 5 percent and eight days at 10 percent replacement. Samples of the fly ash were collected daily from the electrostatic collector hoppers for analysis.

TABLE 1. ANALYSIS OF COAL AND RDF

	Coal as received (percent)	RDF as received (percent)
Moisture	4.34	19.80
Volatile Matter	36.90	N.D.
Fixed Carbon	51.70	N.D.
Ash	7.07	9.10
Heating Value (BTU/lb)	13500	9655
Sulfur	2.40	0.32
Carbon	78.20	37.74
Hydrogen	5.29	5.05
Nitrogen	1.34	0.37
Oxygen	1.39	28.25

	Coal as received (percent)	Burned RDF (percent)
SiO_2	39.87	55.47
Fe_2O_3	22.07	1.94
Al_2O_3	19.13	17.13
TiO_2	1.25	3.20
CaO	4.48	9.48
MgO	1.03	0.35
Na_2O	1.41	4.74
K_2O	1.24	2.80
P_2O_5	1.27	0.54
SO_3	5.43	2.22
MnO_2	N.D.	0.18
NiO	N.D.	0.04
Cr_2O_3	N.D.	0.09
V_2O_5	N.D.	0.41
PbO	N.D.	0.35
Sb_2O_3	N.D.	0.53
BaO	N.D.	0.04
CoO	N.D.	0.04
MoO_2	N.D.	0.42
CdO	N.D.	0.01
LiO_2	N.D.	0.03
As_2O_3	N.D.	0.01
SeO_2	N.D.	0.01

N.D. = Not determined

CHEMICAL ANALYSIS

Chemical analysis was done using Energy Dispensive Analysis X-rays (EDAX) for all metals except the trace elements of lead and cadmium. EDAX samples were mounted on carbon stubs with carbon paste and coated with carbon in a Hummer V sputter coater. The samples were analysed with an accelerating voltage of 20 KV, a specimen angle of 30 degrees at a 12 mm working distance on a general area of the sample at 20X magnification. The sample was analysed until a minimum of 40000 counts were obtained. Lead and cadmium were analyzed after the sample had been dissolved in hydrofluoric acid and boric acid, using a Jarrell Ash Model Atom Scan 2000, Sequential Plasma Spectrometer (2, 3).

PHYSICAL PROPERTIES

The fly ash was tested for its physical properties for use as a pozzolanic material in concrete. The tests were done according to ASTM C 311-77 procedures (4). Eight batches of concrete mixes utilizing coal-RDF fly ash were made: three mixes replacing cement at 10%, 20% and 30%, three mixes replacing sand at 10%, 20% and 30%, and two control mixes (without fly ash). The control mix was designed for a 4000 psi compresssive strength with a water-cement ratio of 0.464, a slump of 4" +/- 1/4" and an air content of 6%. A Type II portland cement manufactured by Atlantic cement was used with crushed glacial gravel and glacial sand from Manchester, New Hampshire. The coarse aggregate was a combination of 25 percent 3/8 inch and 75 percent 3/4 inch aggregates. The cement and sand replacement mixes were made with the 10 percent coal-RDF fly ash. The plastic and hardened properties of the mixes were analysed.

LEACHING TESTS

Leaching tests were done on coal fly ash, coal-RDF fly ash, portland cement concrete, coal fly ash concrete and coal-RDF fly ash concrete. The tests were done using an upflow column (3, 5). The columns used were modified Buchner type funnels. An 80 gm sample of coal fly ash and coal-RDF fly ash was used in each column. The fly ash concretes were broken after 28 days of moist curing and pulverized with a mortar and pestal. A 150 g sample was used for the test which consisted of 75 g passing a No. 10 sieve (10 mm) but retained on a No. 30 sieve (600 um) and 75 g passing the No. 30 sieve. An acid rain comprised of 5.24×10^{-5} N H_2SO_4 and 1.98×10^{-5} N HNO_3 with a pH value of 4.51 was used as the leachant at a flow rate of 0.25 mL/min. Leachants were collected every 24 hours for 14 days. The metal concentrations were determined using a Jarrell Ash Atom Scan 2000, Sequential Plasma Spectrometer.

RESULTS AND DISCUSSIONS

The chemical properties of coal-RDF fly ash presented in Table 2 show small differences between coal fly ash and coal-RDF fly ash as expected. The percent silicon decreases while the percent aluminum remains the same. Iron content decreases at 10 percent replacement. The sum of these three components are greater than the required 70 percent by ASTM C-618. The percent magnesium decreases with an increase in RDF. A very high increase in sulfur is shown with the average of the eight 10 percent replacement samples (standard deviation was 1.32). The 5 percent replacement has a sulfur content about equal to coal fly ash. The loss on ignition increases as the percent RDF increases.

TABLE 2. CHEMICAL PROPERTIES OF COAL-RDF FLY ASH

	Coal Fly Ash	5% Coal-RDF Fly Ash	10% Coal-RDF Fly Ash
Calcium (%)	8.14	8.25	8.55
Magnesium (%)	0.44	0.40	0.22
Silicon (%)	28.12	26.66	24.77
Aluminum (%)	17.77	17.26	17.15
Iron (%)	38.23	38.36	36.21
Sulfur (%)	3.97	3.73	6.15
Sodium (%)	0.43	0.36	0.59
Pottasium (%)	3.73	3.26	3.72
Loss of Ignition (%)	3.14	3.99	5.62
Moisture Content (%)	0.47	0.64	0.39
Trace Elements			
Cadmium (ppm)	3.29	5.08	7.76
Lead (ppm)	75.20	102.55	198.28

Tests on cadmium and lead show that these elements increase as percent RDF increases. The 10 percent replacement increased more than 100 percent in both of these elements as compared to coal fly ash. An E.P. Toxicity test done on the fly ash by the Public Service of New Hampshire indicated that the fly ash was non-hazardous.(6) However, the EPA proposed new test (TCLP) shows several samples to contain hazardous levels of lead.

The physical properties of coal-RDF fly ash presented in Table 3 show increases in the pozzolanic activity index, the water requirement and air requirement. Fineness decreases while soundness (autoclave expansion/contraction) and drying shrinkage were about the same. The increase in pozzolanic activity index implies a more reactive pozzolanic material. An increase in air requirement show more admixture is needed to obtain the proper air content in concrete. Even though there was a decrease in the fineness, the difference was small and the fineness of the fly ash was well within the requirements of ASTM C-618.

TABLE 3. PHYSICAL PROPERTIES OF COAL-RDF FLY ASH

	Coal Fly Ash	5% Coal-RDF Fly Ash	10% Coal-RDF Fly Ash
Fineness (percent retained on # 325 sieve)	4.00	6.28	7.68
Pozzolanic Activity Index, (%)	87.60	102.17	94.19
Water Requirements (%)	91.94	95.30	97.68
Autoclave Expansion(+) or Contraction(-), (%)	-0.04	-0.04	-0.04
Drying Shrinkage, (%)	0.01	0.03	0.01
Specific Gravity, (MG/m3)	2.67	2.78	2.75
Air Requirement, (ml for 18% air)	3.44	3.61	4.43

Table 4 shows the plastic and hardened properties of coal-RDF fly ash concrete and the control. When compared to the control concrete, the slump of the coal-RDF fly ash increases. The air content decreases as percent replacement increases. These properties are common to fly ash concretes. The unit weight shows a slight increase due to the decrease of entrained air content. The relative yield was slightly less in concrete produced by replacing cement with ash and a little higher in concrete produced by replacing sand.

TABLE 4. THE COMPRESSIVE STRENGTH OF COAL-RDF FLY ASH CONCRETE

	Water/ Cementitious Ratio	Actual Water/ Cement Ratio	Slump (ins.)	Air Content (%)	Unit Weight (pcf)	Relative Yield	Compressive Strength (psi)		
							7 Days	28 Days	15 Weeks
Control	0.464	0.464	5.00	5.5	144.87	1.03	3948	5658	6121
10% Replacing cement	0.464	0.515	7.25	3.8	146.09	1.02	3805 *(3550)	5288 *(5033)	5944 *(5689)
20% Replacing cement	0.464	0.580	5.75	2.7	148.02	1.01	3737 *(3317)	5257 *(4837)	6227 *(5807)
30% Replacing cement	0.464	0.663	5.25	2.3	145.88	1.02	2932 *(2452)	4632 *(4152)	5888 *(5408)
Control	0.464	0.464	6.75	4.7	146.45	1.02	4072	5141	5702
10% Replacing sand	0.464	0.514	8.25	2.3	147.66	1.02	4006 **(3646)	5092 **(4732)	5952 **(5592)
20% Replacing sand	0.464	0.564	9.00	1.6	147.46	1.03	3683 **(3218)	4759 **(4294)	5942 **(5477)
30% Replacing sand	0.464	0.614	8.75	1.36	146.45	1.05	3466 **(2971)	4465 **(3960)	5810 **(5315)

* Corrected to 5.5% air content
** Corrected to 4.7% air content
Assume 1% air reduces 150 psi of strength

The compressive strength of the coal-RDF fly ash concrete was lower than the control at 7 and 28 days but at the age of 15 weeks, the strength was comparable for the 10 and 20 percent replacements. The actual water cement ratio increases due to the replacement of cement with fly ash because the water cementitious ratio was held constant in the test concretes. The increase in water cement ratio in fly ash concrete decreases the compressive strength at early ages until the pozzolanic reaction from the fly ash developes.

The results from the leaching test show cadmium and lead did not leach out of the portland cement and fly ash concretes. The concentrations of the metals obtained were below detection limits. The 0.026 ppm detection limit was defined as three times the standard deviation of the blank which in this case was the synthetic acid rain. The leachates from coal fly ash and coal-RDF fly ash show the concentration of lead to be lower than the detection limit, but the concentration of cadmium was higher in the leachates during the first two days. The values obtained were 0.093 ppm and 0.041 ppm for coal fly ash and 0.486 ppm and 0.057 ppm for coal-RDF fly ash for the first and second day respectively. The results obtained were not unusual since the E.P. Toxicity test on the fly ash had already show low metal concentrations.

CONCLUSIONS

The fly ash obtained by burning RDF with coal, up to 10 percent replacement by BTU, was shown to have pozzolanic, chemical and physical properties within the specification of ASTM C-618. The properties of the concrete made with the coal-RDF fly ash follows the trend of coal fly ash concrete. Metals were not leached out during the leaching tests. Fly ash produced from burning coal with 10% or less RDF can be effectively utilized in concrete.

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TECHNOLOGIES FOR MINIMIZING THE EMISSION OF NO_x FROM MSW INCINERATION

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ABSTRACT

It has been known for some time that emissions of oxides of nitrogen (NO_x) emanate from MSW incinerators as from all sources of fossil-fuel combustion. Previously, understanding the formation of and reducing dioxin/furan emissions has held the attention of regulators, industry, and the environmental community, and only recently has the focus turned to the importance of and methodologies for minimizing NO_x emissions. This paper examines the formation and environmental impacts of NO_x and the four general categories of NO_x reduction methodologies (source separation, combustion modification, add-on devices, and operations). Also described are several of the commercially successful NO_x reduction designs and devices. In addition, NO_x emissions data from recent tests on MSW incinerators equipped to minimize NO_x and the current regulatory trends as regards NO_x emissions are presented and discussed. The data support the probability that implementation of a combination of these technologies could result in NO_x emissions under 100 ppm.

INTRODUCTION

NO_x or oxides of nitrogen, are produced in incinerators in the forms of nitric oxide (NO), the predominant form of NO_x produced, and NO_2 (nitrogen dioxide) which is also generated in small amounts. In the presence of oxygen NO is converted to NO_2, which contributes a brownish red cast to the atmosphere, reducing visibility, especially at the horizon. NO_x in the atmosphere is also both a precursor to ozone formation and acid fog and rain.

Oxides of nitrogen (or NO_x) are produced from two separate mechanisms. NO_x is produced both due to the combustion of nitrogen contained in the waste itself (referred to here as fuel NO_x) and due to the interaction at high temperature of atmospheric nitrogen with the available oxygen. The product of the latter process (common to all types of combustion) is referred to as thermal NO_x. In addition to oxygen availability, the amount of thermal NO_x generated is a function of the furnace heat release rates and residence time at the peak temperatures. Specifically, the factors influencing production of fuel NO_x in an incinerator include the distribution of the combustion air (underfire vs. overfire), the fuel nitrogen content, and total excess air rates. Thermal NO_x formation rates increase with temperature at temperature exceeding 1600°F, oxygen availability, heat release rate and flue gas residence time at high temperatures.

The relative contribution of fuel and thermal NO_x to the total NO_x emitted from an incinerator is dependent on the design and operation of the furnace and the nitrogen content of the refuse burned. At temperatures less than 2000°F, typical of MSW incinerators, NO_x emissions are influenced mainly by oxidation of fuel nitrogen. Thus, generally, 75 to 80 percent of the total NO_x emitted from incinerators may be fuel NO_x (Ref. 1).

NO_x CONTROL METHODS

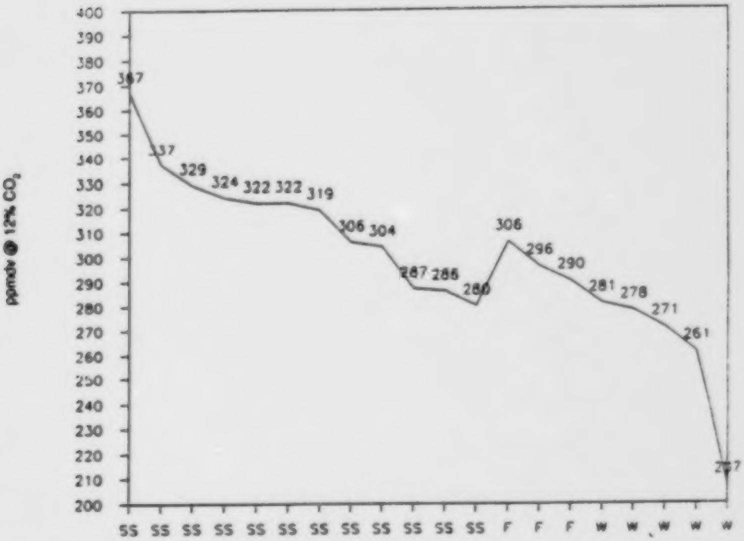
Nitrogen oxides from solid waste incinerators can be reduced by four methods: reducing the nitrogen content of refuse introduced into the furnace (source separation of yard waste, for example), minimizing the amount of oxygen available for combustion with available nitrogen at the point of combustion (combustion air modifications and flue gas recirculation, for example), post-combustion reduction of the quantity of NO_x in the flue gas stream (flue gas controls) and operating techniques to enhance the effectiveness of combustion design modifications and flue gas controls

SOURCE SEPARATION

Grass clippings, leaves and other yard waste contribute significantly to the nitrogen content of residential solid waste, and food wastes are also high in nitrogen content. This being the case, and since fuel NO_x is the primary form of NO_x created in incinerators, it appears that separate collection of yard wastes for the purpose of composting would not only have a high potential for reductions in NO_x emissions from incinerators but would also be cost-effective, since yard wastes are already separated from other refuse by the homeowner. In addition, since yard waste contributes roughly 20% to the volume of solid waste generated in this country, instituting yard waste composting on a large scale would have the additional benefit of reducing the specialized capacity (e.g. state-of-the-art incinerators, landfills, ashfills, etc...) otherwise needed for processing and disposing of unprocessed MSW.

Research to quantify the effect of the contribution made by yard waste to NO_x emissions from MSW incinerators has only recently begun. One way to demonstrate the relationship between yard waste and NO_x emissions would be to observe the effect on NO_x emissions associated with the normal variation in the percentage of yard waste in the waste stream. Since the production of grass clippings (which generally accounts for the bulk of yard waste on an annual basis) is highest in the spring and summer months (or when there is the most sunlight, warm temperature, and ample rain), and lowest in the winter months when growth ceases, and since yard waste accounts for an average 20% of the total waste stream, normal seasonal changes would account for a marked variation in the nitrogen content of MSW.

To demonstrate this effect, a study of NO_x emissions from a number of modern incinerators during different times of the year was undertaken by Hahn. The facilities used in the study included Marion, Hillsborough, Alexandria, Bristol, Stockholm, Tulsa and Wurzburg; none of these facilities employ special NO_x reduction devices. As expected, a rough correlation was found between the concentration of NO_x emitted (usually as a weekly average of several tests) and the season (Ref. 2). Figure 1 illustrates the findings graphically. The range of NO_x emissions measured in the spring and summer months was 280 to 367 ppm while the range measured in the winter months was 207 to 281 ppm (all



SOURCE: "Variability of NO_x Emissions From Modern Mass-Fired Resource Recovery Facilities" Jeff Hahn and Donna Sofaer. APCA Annual Meeting, Dallas, TX June 19-24, 1988.

SS = Spring/Summer
F = Fall
W = Winter

Figure 1. Seasonal Variations in NO_x Emissions

values corrected to 12% CO_2). Emissions in the fall months overlapped the spring/summer values to a minor extent (290 to 306 ppm), though it should be noted that the highest fall reading (306 ppm) was measured during a warmer than usual October, and the lowest spring/summer reading (280 ppm) was measured in Stockholm in late August, which might have been cooler than the other locations during the late summer. Measurement of the average weekly NO_x emissions from a number of plants vs. the average weekly temperature, or vs. a temperature/ rainfall/ sunlight factor at the time and place of NO_x measurement, or direct measurement of fuel nitrogen would probably provide a more accurate relationship between fuel NO_x and incinerator NO_x emissions irrespective of the climatic zone, weather patterns, season, and latitude of the test facilities during the test period. Conclusive demonstration that separation of yard wastes from MSW to be incinerated reduces NO_x emissions, particularly in the spring and summer months when ozone is more of a concern, would be significant, especially since this option is relatively less expensive to implement and does not require retrofitting.

COMBUSTION DESIGN MODIFICATIONS TO MINIMIZE NO_x

As is the case with other pollutants, combustion design modifications help to reduce formation of NO_x in the incinerator. Five combustion designs/operating strategies which effect control of fuel and thermal NO_x are

(1) grate and furnace design for staging of combustion,

(2) waterwall cooling of the grate area,

(3) overall boiler design to permit flue gas recirculation,

(4) bubbling and circulating fluidizing bed boilers, and

(5) automatic combustion controls.

Low Oxygen Conditions in the Furnace / Staged Combustion

A lower excess air (or starved air) condition at the point of combustion of MSW minimizes the potential for NO_x formation by reducing the oxygen available for reaction with the available nitrogen and by limiting the temperature that can be achieved in the primary combustion chamber. The staging of combustion in a dual-chambered incinerator is the typical incinerator design to accomplish a low stoichiometric oxygen condition to minimize NO_x formation. In this arrangement, the primary chamber usually operates in a starved air mode with excess air in the secondary chamber. Since a low oxygen condition is maintained in the primary chamber, less fuel NO_x is created, though PICs are formed. The excess air in the secondary chamber is designed to complete the combustion process.

The staging of combustion is approximated in a typical mass burn incinerator, in effect, by systems which regulate the distribution of air to underfire and overfire air injection systems, encouraging fuel-rich combustion at the grate and air-rich combustion above the grate where overfire air is injected.

Boiler designs which promote this condition include:

o Systems to regulate distribution of underfire air among several injection zones under the grates (zone control). Underfire or primary chamber air injection zones can be operated to supply the optimal amount of oxygen for minimizing NO_x generation (usually an oxygen poor condition), vis a vis the quantity and BTU content of the waste on the grates while at the same time not adversely affecting combustion efficiency. This is accomplished if the low oxygen at the primary stage is used in concert with overfire air injection or a secondary chamber where additional air is made available to complete combustion of the gases once they pass out of the region of the grates. Many modern incinerators employ zone control to regulate combustion in the furnace.

- Incorporating in the design the optimal number of grate holes for injecting underfire air. This design permits less underfire air (less oxygen) to be injected (though at a greater velocity) while minimizing plugging of the air holes with refuse.

A well-tested example of a dual-chambered controlled-air incinerator is the Prince Edward Island facility (Ref. 3). During testing the primary chamber temperature was maintained at a relatively constant 1292°F -- within 104°F, except for the low temperature test, where it was maintained at 1249°F. Secondary chamber temperatures were kept at 1544°F for two tests, at 1922°F for the high temperature test, and 1364°F for the low temperature test. The NO_x emissions were 200 and 170 ppm (uncorrected) for the first two tests, 220 ppm for the high temperature test, and 160 ppm for the low temperature test, thus low primary chamber temperature and limited secondary temperature, and the smoldering nature of the fire in primary chamber as observed during the tests, which is generally associated with low oxygen, correlates well with the lower NO_x emissions, at least when the emissions are not corrected to 7% O_2.

A third type of staged combustion for NO_x control, developed by Nippon Kokan in Japan, is a split chamber grate furnace. The split chamber, somewhat similar to the above dual-chambered incinerator, divides the flue gas in fuel-rich and air-rich streams (40% and 60% respectively). The fuel-rich stream reaches a maximum temperature of 1652°F and contains essentially no NO_x, 8-10% CO, and devolatilization products. The air-rich stream contains about 80 ppm NO (at 12% O_2) and reaches a maximum temperature of 1760°F. When the two streams recombine, the result is less than 50 ppm NO (12% O_2) or 75 ppm corrected to 7% O_2. Though NO_x emissions are low, no specific information about the effect of lowering the furnace temperature to 1650 to 1760°F is available for this type of staged combustion unit other than "dioxin emission measured from the furnace is low" (Ref. 4). A possible explanation is that the residence time of the combustion gases in the split furnace is higher allowing for an improved combustion efficiency. (This is the case for fluidized bed systems which typically operate at 1500°F and 5 seconds residence time and achieve low NO_x and low dioxin emissions.)

Rotary Waterwall Mass Burn Design

Another example of a furnace design which limits the flame temperature is the Westinghouse/O'Connor rotary waterwall design. The combustion of solid waste takes place in a slightly inclined, cylindrical, rotating furnace of waterwall design which enhances agitation of the waste while limiting the flame temperature due to the cooling effect of the waterwall. (Typical mass burn furnace grates are not water-cooled, resulting in higher flame temperatures.) This technology is operated at 50% excess air rather than the 100% excess air typical of mass burn facilities. Though only preliminary data is available, average CEM NO_x levels from both Bay County, Florida and Dutchess County, New York rotary combustors indicate emissions in the 150 ppm range, and it is theorized that the waterwall design and the low excess air operation both contribute to this emission level (Ref. 5).

Fluidized Bed Technology

Both forms of fluidized bed technology ("bubbling" and "circulating") are types of combustion that have been demonstrated to produce relatively lower levels of NO_x emissions. Table I indicates that the Duluth Bubbling Fluidized Bed plant emits 24 ppm of NO_x and the Sundsvall Gotaverken Circulating Fluidized Bed plant emits 91 ppm. These lower NO_x emissions (vs. over 300 ppm for uncontrolled emissions) are probably due to the fact that fluidized bed plants combust solid waste via intimate mixing for comparatively long residence times (about 5 seconds) at temperatures in the vicinity of 1500°F, well below temperatures of typical large, mass burn excess air designs (1800 to 2000+°F) and thus, minimize thermal NO_x production.

Automatic Combustion Controls

A recent example of automatic combustion controls resulting in reduced NO_x emissions from a resource recovery facility is in use in the Jackson County, Michigan mass burn, waterwall, reciprocating grate plant. To achieve a lax standard of 311 ppmdv NO_x (12% CO_2) the plant was designed with automatic combustion controls. At full load, an average of three tests indicated 156 ppmdv NO_x emissions from Unit 1 and 150 ppmdv from Unit 2 (Ref. 6).

Table I

NO_x EMISSIONS ACHIEVED BY NO_x PREVENTION AND REDUCTION TECHNOLOGIES

Systems/Plants	**Emissions level Attained or guaranteed @ 12% CO_2 or 7% O_2**	**Remarks**	**Reference**
Uncontrolled (spring/summer)			Hahn. APCA 88
Marion	286 ppm } avg. 326		
Tulsa	367 ppm		
Hillsborough	325 ppm		
Source Separation (winter)			Hahn. APCA 88
Alexandria	207 ppm } avg. 249		
Bristol	280 ppm		
Wurzburg	261 ppm		
Combustion Control			
Red Wing, MN	135 ppm	test March. 88	Interpoll
Automatic Combustion Control (ACC)			
Hitachi Zosen., Japan	140 - 187 ppm		vendor lit.
Quebec City retrofit	190 ppm		Env. Canada
Jackson, MI	150-156 ppm	test late April. 88	Research Cottrell
Olmsted, MN	187 ppm	test late September. 88	Riley-Takuma
Water Spray in Furnace			
Hitachi Zosen. Japan	125 - 155 ppm	used w/ACC	vendor lit.
Low Excess Air/Grate Cooled			
Bay County, FL	150 ppm avg.	preliminary data	Westinghouse
Dutchess County, NY	150 ppm avg.	preliminary data	Westinghouse
Controlled Air/Staged Combustion			
Oneida, NY	90 ppm		NYSDEC
Flue Gas Recirculation			
Hitachi Zosen	93 - 125 ppm	used w/ACC	vendor lit.
Rutland VICON (Performance test)	90 ppm	used w/ACC	VICON
Rutland VICON	60 ppm	w/condenser & wet caustic scrubber. ACC	VICON
Kita, Japan VOLUND	60 - 85 ppm		CARB
Pittsfield VICON	100 - 150 ppm	with ACC	NYSERDA
Rutland VICON (Compliance)	152 ppm	Commercial Waste	ENERCON
Springfield, MA VICON (Compliance)	143 ppm	Commercial Waste	ENERCON
SNCR (Thermal DeNO$_x$)			
Hitachi Zosen	78 - 125 ppm	used w/ACC	vendor lit.
Commerce, CA	90 ppm	used w/ACC	Wheless
Bremerhaven, W. Ger.	37 ppm	retrofit: inlet ~160 - 215 ppm. used with wet scrubber and ammonia stripping	Von Roll
SCR			
Hitachi Zosen	62 - 93 ppm		vendor lit.
Urea Injection — NO$_x$ OUT			
Emmenspitz. Switz.	60-80 ppm	retrofit: inlet ~200 ppm	Fuel Tech
Wet Scrubber			
Helsingborg, Sweden	95% removal	AOCS w/condenser	Raring Corp.
Hitachi Zosen	62 - 93 ppm	w/oxidation	vendor lit.
Circulating Fluidized Bed			
Sundsvall, Sweden	91 ppm		Gotaverken
Bubbling Fluidized Bed			
Duluth	24 ppm	w/venturi & tray scrubber	Interpoll

Flue Gas Recirculation

Flue gas recirculation systems, which, like underfire/overfire air systems, can also be retrofitted into incinerators, involve recycling of relatively cool flue gas containing combustion products and lower oxygen concentrations into the furnace, thereby lowering NO_x formation by limiting the flame temperature and slightly diluting the flame oxygen concentrations. Flue gas recirculation is primarily used as a means to lower thermal NO_x but when circulated to the primary air zone this technique can aid in improved staging of the combustion process and reduce fuel NO_x as well (Ref. 1).

Flue gas recirculation tests conducted at the Kita incinerator in Tokyo, a Volund refractory wall furnace, resulted in a 25% reduction of NO_x emissions by recirculating the flue gas equivalent to 20% of the combustion air. The NO_x emissions there ranged from 70 to 80 ppmv (dry 12% CO_2). Other full-scale applications of this technology are at the Copenhagen Amager and Copenhagen West incinerators, where flue gases at 570-660°F are recirculated to the furnace from the exit of the ESPs with successful results (Ref. 1).

The ENERCON technology also employs flue gas recirculation on its plants along with a dual-chambered automatic combustion control system. In this system a thermocouple controls the amount of air recirculated to the furnace, and an oxygen meter controls the fresh air brought into the system. At the Pittsfield facility which was tested repeatedly (19 runs) for NO_x as part of the ASME combustion research project, the NO_x emissions were generally between 100 and 150 ppm (corrected to 7% O_2) (Ref. 7). During performance tests, the Rutland, Vermont facility produced 90 ppm of NO_x prior to the condenser/scrubber and 60 ppm after (Ref. 8). However, during Rutland's subsequent compliance testing where a high percentage of commercial wastes were burned (producing hotter furnace temperatures), NO_x emissions were 152 ppm (12% CO_2). Similarly, the new Springfield, MA ENERCON facility compliance tests on a relatively large proportion of commercial waste indicated emissions of 143 ppm NO_x (Ref. 9).

OPERATIONS TO MINIMIZE NO_x PRODUCTION AND EMISSIONS

Since generation of NO_x depends to a large degree on the availability of nitrogen and oxygen in the flue gas as well as the temperature, an increase of these factors serves to increase

NO_x emissions. Two of the combustion technologies to control fuel and thermal NO_x mentioned above as facility design modifications (low excess air operation and staging of combustion) also involve specific operations to minimize generation of NO_x. To minimize emissions of NO_x in plants where there are no automatic controls in order to keep furnace temperature in a desired range and modulate either the distribution of fresh and/or recirculated air among the underfire air injection zones or the distribution between underfire and overfire air (or between chambers in modular systems), the operator must effect manual control of both air distribution and furnace temperature. Since a lower oxygen condition just above the grates (or in the primary chamber in dual chambered facilities) coupled with a higher excess oxygen condition where overfire air is injected (or in the secondary chamber in dual chambered facilities) is recommended for minimizing NO_x production, the operator in this case would manually adjust the stoker speed rate and the primary combustion air feed rate to provide the least amount of underfire air that would not result in plugging the injection holes in the grates. In fact, a test of the effects of low excess air and staged-combustion techniques by Hiraoka over a nine-month period indicated that NO_x emissions could be maintained at less than 220 ppm (12% CO_2) by manual adjustment of primary air and stoker speed, and that utilizing an automatic control system reduces NO_x another 33% to 140 ppm (Ref. 1 and 10).

NO_x EMISSION CONTROL DEVICES

As was described above, source separation and combustion modifications help to reduce production of NO_x during the combustion process, but formation of some NO_x from whatever nitrogen remains in the fuel post-separation and from atmospheric nitrogen is inevitable, necessitating use of flue gas controls to achieve any further reduction of these emissions. Four of the most promising NO_x flue gas control systems under development at present are selective catalytic reduction (SCR), two selective non-catalytic reduction (SNCR) technologies - Thermal $DeNO_x$, and Urea Injection (e.g. NO_xOUT), and wet flue gas denitrification (FGD_n), though the latter technology suffers, despite its purported high efficiency, due to the waste water effluent problem and high construction and maintenance costs (Ref. 11). Though Thermal $DeNO_x$, NO_xOUT, and SCR are in the early stages of commercialization for use on resource recovery plants, some early pilot studies and test results on MSW incinerators are promising and data from use on other applications indicate removal efficiencies of 60 to 90% (Ref. 1, pp. 99-102).

Selective Non-Catalytic Reduction

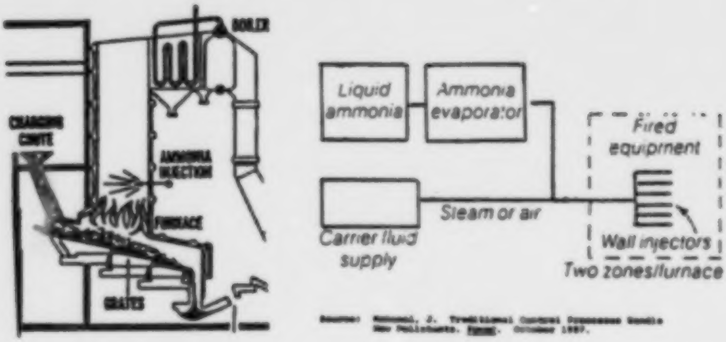
Since NO_x emissions impede visibility and aggravate already high levels of ozone in southern California and other urbanized areas (and also contribute to the formation of acid rain), several facilities equipped for SNCR of flue-gas NO_x have been planned there. Selective Non-Catalytic Reduction, of which Thermal $DeNO_x$ is one type, achieves NO_x reduction by injection of ammonia (NH_3) into the furnace at about a 2:1 to 3:1 ratio of NH_3 to NO via large jets positioned on the furnace walls at a location where temperatures are in the vicinity of 1600 to 1800°F. (Other forms of SNCR may use aqueous ammonia or urea solution in place of ammonia as the chemical agent). In the proper temperature range for Thermal $DeNO_x$ the injected ammonia combines with NO to form water vapor and elemental nitrogen. Above 2000°F injection of ammonia becomes counterproductive, causing additional amounts of NO to be formed, and below 1600°F ammonia breakthrough (a visible ammonium chloride plume) can occur (Ref. 1, p. 99).

At the Commerce plant the Thermal $DeNO_x$ system has achieved a relatively low NO_x emission concentration of 90 ppm (corrected to 7% O_2) with the Thermal $DeNO_x$ operating at normal levels. At a high ammonia injection rate, a NO_x emission level of one-third lower than this figure was achieved (60ppm). Insignificant (less than 2-3 ppm) ammonia breakthrough occurred, presumably due to careful control of furnace temperatures, molar ammonia injection ratios, and/or combustion stability (Ref. 12). By comparison, with the thermal $DeNO_x$ off, NO_x emissions were almost twice that with $DeNO_x$ operating at normal levels, a 50% reduction efficiency for the $DeNO_x$ system.

Exxon's Thermal $DeNO_x$ is currently being licensed from the Mitsubishi Heavy Industries of Japan where it was originally developed and demonstrated on numerous applications. At one such equipped plant in Japan a NO_x removal efficiency of 80% and emission concentration of 20 ppm was achieved using this technology (Ref. 4).

It has been found that Thermal $DeNO_x$ operates most successfully when combustion conditions are kept under control and fluctuations in temperature and oxygen conditions are kept to a minimum. Feedback systems, which sense the normal fluctuations in NO_x created as a result of the considerable variation in the quantity and BTU value of refuse on the grate at any given time, and which stabilize combustion or steam flow rate by automatically adjusting the underfire air have been found to be reliable in minimizing the large fluctuations in NO_x production, thus maximizing the removal efficiency achieved by the thermal

DeNO$_x$ system while also minimizing the ammonia breakthrough (Ref. 11).


BOILER
CHARGING CHUTE
AMMONIA INJECTION
FURNACE
GRATES
Liquid ammonia
Ammonia evaporator
Fired equipment
Steam or air
Carrier fluid supply
Wall injectors
Two zones/furnace

Source: Makansi, J. Traditional Control Processes Handle New Pollutants. Power. October 1987.

Figure 2. General and Detailed Schematic of the Thermal DeNO$_x$ Process

Thermal DeNO$_x$ can be retrofitted into older furnaces, but since the location of the injection system is critical to achieving optimum performance and NO$_x$ removal efficiency, sometimes disallowing selection of the optimal location for injection, a cooler location may be selected for ammonia injection and hydrogen simultaneously injected to extend the process' effectiveness to lower flue gas temperatures (Ref. 13).

An example of such a retrofit into an older Von Roll incinerator took place at the Bremerhaven facility in West Germany. A Selective Non-Catalytic Reduction device has been operating successfully since January, 1988 in tandem with a Von Roll system to recover ammonia from the scrubber liquor. (The latter works since ammonia is highly soluble in water, and is thus collected with over 95% efficiency by the wet scrubber in use there -- inlet value 100 mg/Nm3; outlet value below 5 mg/Nm3.) This system operates by stripping ammonia from the scrubber liquor with steam and subsequent reinjection into the boiler. The total NO_x removal efficiency amounts to 80% considering an average inlet concentration of 350 mg/Nm3 and maximum outlet concentration 70 mg/Nm3 (Ref. 14).

Urea Injection -- NO_xOUT

A newer SNCR technology for post-combustion reduction of NO_x in the early stage of commercialization on incinerators is NO_xOUT, a chemical process originally developed during research in the late 1970s sponsored by the Electric Power Research Institute (EPRI). This system, which was subsequently commercialized on fossil-fuel plants and patented by Fuel Tech, is easily retrofitted into existing combustors. In the NO_xOUT process, an aqueous solution containing urea and/or chemical enhancers is injected into the furnace and boiler at one or more locations, depending on type and size of combustor. These reagents react with NO_x converting it to nitrogen, carbon dioxide, water vapor, and small amounts of ammonia. An advantage of using urea vs. ammonia as the primary reagent is that the former is nontoxic, whereas the latter is a toxic compound and its production, use, storage, and transport can be hazardous, and its use is thus tightly regulated by OSHA, EPA, and DOT (Ref. 15, p. 4-38). Another advantage is the greater efficiency of mixing of reagents with NO_x achieved by injecting an aqueous solution (with urea) vs. a gas (ammonia).

The NO_xOUT process provides the greatest degree of NO_x control and minimum ammonia breakthrough when:

(1) the chemical mixture (urea plus proprietary additives) has a high activity for NO_x reduction at the treatment temperature (i.e. the temperature of combustion gases at the point of injection),

(2) there is intimate mixing of NO_x and the chemical mixture, and

(3) the proper amount of chemicals is used vis a vis stoichiometric quantities.

At temperatures typical of modern mass burn incinerator furnaces (1700 to 1900 °F), urea alone in water has a high NO_x reduction activity. By using different combinations of chemical enhancers and adjusting their relative concentrations and optimizing the location of injection vis a vis combustion gas temperatures, urea-based mixtures are effective in the range of

1500 to 2100°F. The additives are widely available and do not require special provision for safe storage and handling. Figure 3 illustrates how the use of different process chemicals extends the temperature range at which NO_x is efficiently reduced.

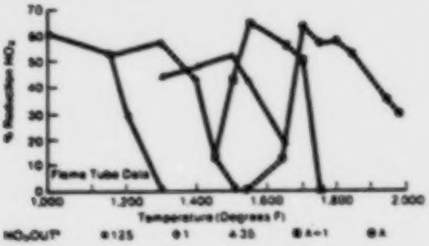


Figure 3. NO_x Reduction vs. Temperature

To improve the degree of mixing attained,

- injectors are designed to give proper penetration based on boiler geometry,
- injectors are spaced to be consistent with spray patterns generated, and
- multiple level (site) injection is used as a means of increasing the degree of reduction and minimizing ammonia breakthrough. (Figure 4 illustrates a NO_xOUT configuration with three levels.)

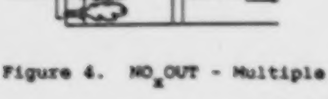


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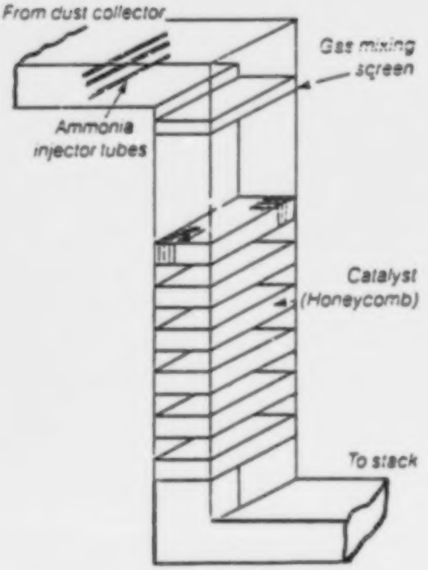
Figure 4. NO_xOUT - Multiple Level Injection

Though NO_xOUT has been demonstrated with a number of fuels (including hard and brown coal, oil, gas, and wood) and combustor types (including, among others, circulating fluidized beds), it has been demonstrated on only one MSW incinerator to date, though efforts are underway for more such demonstrations. This older incinerator, operated at between 1200 and 1800°F, achieved a NO_x removal rate of 68% in this 1988 test (Ref. 16). The uncontrolled NO_x emissions of roughly 200 ppm (a low figure probably due to the low furnace temperatures at this early, 250 TPD Von Roll mass burn plant -- Emmenspitz at Zuchwil, Switzerland) were reduced to between 60 and 80 ppm with the addition of a two-level configuration of NO_xOUT using two chemical additives in addition to urea. (Specifically, the first level was at higher furnace temperature location using urea plus #34 additive which extends the effective range of urea and minimizes ammonia slip; the second level was at lower furnace temperature location where #63 additive was injected alone to provide an additional 10% reduction of NO_x) (Ref. 17). Future planned tests on more modern MSW incinerators are expected to achieve greater levels of NO_x reduction due to the typically higher furnace temperatures (which are more conducive to the urea/NO_x reaction without the need for additives) (Ref. 18).

Selective Catalytic Reduction

Selective catalytic reduction (SCR) also achieves reduction of NO_x emissions by injection of ammonia, but at a different location and by different means. Instead of simple injection of ammonia into the flue gas, SCR consists of ports of permit injection and mixing of the ammonia with the flue gas and a catalyst bed situated downstream. The catalyst material can be shaped into a variety of forms (honeycomb plates, parallel ridged plates, rings, tubes, pellets, etc...) and can consist of a variety of base metals (e.g. copper, iron, chromium, nickel, molybdenum, cobalt, and vanadium). Each of these combinations has advantages and disadvantages with respect to catalyst/NO_x contact, fouling of the catalyst, and pressure drop through the catalyst (Ref. 1).

SCR technology is capable of achieving similar NO_x removal efficiencies to Thermal DeNO_x with use of half the ammonia due to the positive effects of the much lower temperature of reaction and the selective catalyst (Ref. 19). However, due to the fact



Source: Makansi, J. Traditional Control Processes Handle New Pollutants. *Power*. October 1987.

Figure 5. Schematic Of the SCR Process

that SCR operates most efficiently at flue gas temperatures of roughly 570 to 750°F and when the flue gas is relatively free of particulate matter, (this, due to the poisoning effect on the catalyst of particulate in the flue gas), and most particulate control equipment operates at temperatures of less than 600°F, SCR for application in incinerators is usually located at the end of the pollution control train after the flue gas stream has been cooled by passing through an economizer and a particulate collection device. As a result, flue gas reheat (accounting for as much as 8% of the heat released from the furnace) is required to bring the gas temperature up to that required for SCR. Consequently, research is proceeding into the development of low temperature catalysts and catalysts which are relatively insensitive to particulate matter concentrations (Ref. 1, p.94).

One alternative arrangement including SCR is non-heat recovery incinerators has been to position it after a hot-side ESP, where less cooling and reheating would be necessary. The Kawasaki SCR technology is situated in the back end of the emissions control train after the flue gas passes through the spray dryer (flue gas exit temperature 270°F), the fabric filter, induced draft fan, and heat exchanger/supplemental burner (which heats the flue gas up to roughly 600°F). After the flue gas has been heated, it is injected with ammonia (1:1 ratio with NO) and it then passes through a heat exchanger which removes energy while reducing the temperature roughly 370°F before it exits the stack (Ref. 19).

SCR is currently being used on two solid waste incinerators in Japan, including the Iwatsuki plant which also has a baghouse. This plant achieves 50-60% removal of NO_x with less than 10 ppm ammonia slip (the threshold for visibility of the plume) (Ref. 20). An SCR installation is also currently under construction on the Martin Munich-South plant. A pilot SCR was installed on the plant in early 1988 and testing of about a dozen catalysts was undertaken during the year. The full-scale SCR is being fitted with a ceramic honeycomb vanadium catalyst designed for 90% NO_x reduction. Assuming inlet NO_x levels of 350 ppm, emissions are expected to be about 35 ppm at system start-up in 1990 (Ref. 21).

Wet Scrubbing/Condensation

As with the other NO_x removal technologies, the capacity for wet scrubbing/condensation systems to achieve control of NO_x is now being explored. As noted above, the Rutland Condenser/Wet caustic scrubber achieved a 33% reduction (60 ppm) aside from the reductions gained by staged combustion and flue gas recirculation. Preliminary data from the Helsingborg ACGS condenser/wet scrubber indicates a 95% NO_x reduction, and Hitatchi Zozen, a Japanese vendor guarantees NO_x emissions between 62 and 93 ppm.

In summary, there is a wide range of technologies that can be used to reduce NO_x emissions from MSW incinerators to lower levels, used either alone or together with other technologies. Table I shows emissions of NO_x that have been achieved by operating incinerators utilizing the technologies and practices described above (or that have been guaranteed by a vendor).

CONTINUOUS EMISSION MONITORING FOR NO_x

Measurement of NO_x to ascertain instantaneous emission levels may become easier with the use of NO_x Continuous Emission Monitors (CEMs). At the Commerce test, concurrent measurement of NO_x emissions by means of CEM and conventional sampling showed virtually the same measurement (Ref. 12).

EMISSION LIMITS FOR NO_x

At present, a number of maximum resource recovery emission limits for NO_x (ranging from 100 to 300 ppm) have been established as state regulations and permit conditions. These include the 140 to 200 ppm limit set in the California guidelines which were established over five years ago, the SEMASS plant permit (196 ppmv), the Portland, ME permit (100 ppm) and the Crestwood, IL permit (100 ppm). In addition, Tokyo City has a NO_x limit of 109 ppm (corrected to 7% O_2). Recently, New York State incorporated a statewide BACT requirement for NO_x (and LAER in ozone nonattainment areas) for all new incinerators, and similar requirements are being considered for existing incinerators. In New Jersey the Pennsauken permit requires SNCR technology as BACT for NO_x control (Ref. 18). Also, the State of Connecticut considers SNCR as BACT for some new plants (e.g. Lisbon Resource Recovery plant) (Ref. 22). In addition, the State of California has required SNCR for new incinerators for five years, and is considering the additional requirement of SCR on each new plant permitted (Ref. 23). Unfortunately, at this time it is not possible to predict whether NO_x emissions will be regulated on a national level.

However, due to the preoccupation over the past several years with minimizing formation and emission of dioxin/furan, little attention has been focussed on developing technologies or stringent regulations designed to reduce NO_x. As a result, many of the NO_x permit limits and state guidelines in this country were adopted prior to successful demonstration of the various denitrification technologies on resource recovery plants here and abroad, and assume no attempt in the design or operation of the plant to control NO_x emissions.

CONCLUSION

Considering the recent developments in NO_x control technologies and their successful demonstration and testing on new and existing municipal incinerators and resource recovery facilities, it is probable that implementation of a combination of technologies including yard waste separation/composting, combustion modifications, improved operations, and/or add-on NO_x removal devices could result in NO_x emissions well under 100 ppm. Thus, in areas where NO_x, ozone, or acid rain/fog is considered a local or regional problem (e.g. the Eastern Seaboard, California, the Midwest, etc...), future limits for NO_x could be set at lower levels than in the past.

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REDUCTION OF NITROGEN OXIDES WITH SCR - TECHNOLOGY IN SOLID WASTE INCINERATORS

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ABSTRACT

A conventional incinerator for domestic waste produces several relatively harmful substances:

- acidifying gases HCl, SO_2, NO_x
- heavy metals Cd, Pb, Hg
- toxic chlorinated hydrocarbons

The air pollution control system must therefore be equipped with:
- a filsorption (filtration + adsorption) stage for submicron particles and toxic hydrocarbons.
- an absorption stage for the water soluble compounds
- a reduction stage for elimination of the nitrogen oxides

To be able to come down to the same emission levels as for HCl and SO_2 or about 20 ppm, only a selective catalytic NO_x reduction (SCR) system can be chosen.

Catalysts can be deactivated in different ways by heavy metals and reaction products. Therefore the catalysts have to be located downstream of a very effective gas cleaning stage. As zinc and then particulate zinc chloride is regarded as a strong catalyst deactivator the very best equipment for removal of submicron dust must be included.

For the Munich South plant it was decided to install a Conditioned Dry Absorption System (CDAS) with fabric filter (FF) followed by a SCR stage to reach the requested emission levels (HCl < 6, $SO_2 < 9$, $NO_x < 35$ ppm).

In order to choose the optimal catalyst, a pilot SCR $DeNO_x$ plant has been operated downstream of the CDAS/FF plant in Kempten Bavaria. Three catalysts have been tested at 320 °C (610 °F) of 2 300 hours. They have all worked excellently with high NO_x reduction, very low ammonia slip and no deactivation. The combination Dry scrubber CDAS/FF and SCR $DeNO_x$ - has proven to be an excellent APC-system for MSW-Incinerators.

BACKGROUND

Combustion of wastes produces a number of relatively toxic compounds;

- acidic gases
  - hydrogen chloride - for domestic waste approx. 600 ppm
  - sulphur oxides - " - 100 ppm
  - hydrogen fluoride - " - 10 ppm
  - nitrogen oxides - " - 200 ppm
- heavy metals
  - particulates - such as cadmium, lead, zinc.
  - gaseous - mainly mercury.
- toxic hydrocarbons such as chlorinated benzenes, phenols, dioxins and furanes.

ACIDIFYING GASES

In modern incinerators with effective flue gas cleaning the emissions of sulphur dioxide and hydrogen chloride are reduced considerably before entering the stack; typical levels are in the range of 25 ppm. New regulations are proposed for instance in FRG, Federal Republic of Germany, which will decrease this level further to approximately 10 ppm.

Nitrogen oxides are acidifying gases as well as SO_2 and HCl. Further to this negative property they also work as nitrogen fertilizer which is damaging to soils, lakes, rivers etc.

Waste incinerators do not emit too high a level of NO_x, but calculated over a year they often have a rather high total emission due to their long operation time. The specific emitted amount of NO_x per produced usable energy unit (g/MJ) is also rather high as the waste is a low calorific fuel and the combustion is often performed with a high excess of air and with a low efficiency.

Consequently there is no choice, we have to reduce the emission of nitrogen oxides.

DENO$_x$ - METHODS

Combustion of domestic wastes is always performed at a comparatively low temperature or below 1 000 °C (1 800 °F), therefore most of the NO$_x$ is coming from the nitrogen in the fuel.

NO$_x$ can be reduced in a number of ways (ref 1):

- reduction of nitrogen in fuels
- optimization of combustion
- selective non catalytic reduction (SNCR)
- selective catalytic reduction (SCR)
- wet and dry scrubbing methods

As the reduction of nitrogen oxides must be in the same order as for HCl and SO$_2$ or down to approximately 20 ppm, only selective catalytic reduction with ammonia is possible to apply here.

SCR - DENO$_x$

Selective catalytic reduction with ammonia over, for instance a V_2O_5 - TiO_2 - catalyst, is a proven technology for natural gas, heavy oil and many kinds of coal (ref 2).

As the catalytic reaction is a surface phenomenon it is of course very important that the catalyst does not become deactivated. The worst "poisons" for the catalyst in the case of incinerator are heavy metals, particularily zinc and then zinc chloride

SCR with a location directly downstream of the incinerator boiler has been tested (ref 3), with very bad results. At the same time a location downstream of an electrostatic precipitator (ESP) operating at high temperature (approximately 270 °C) (520 °F) was also tested giving promising results and some full scale installations are in operation in Japan with this concept or similar e.g. fabric filter (FF) instead of ESP - a so called "low dust" location.

ESP or FF operation at these high temperatures often produces dioxins and furans (ref 4) therefore a temperature of 220 °C (430 °F) or lower is necessary for the dust collector.

Most of the heavy metals including zinc and zinc chloride are concentrated to the submicron particles therefore the very best collector for this dust must be installed. SO$_2$, SO$_3$ and HCl all react with ammonia at low temperature therefore it is of great advantage if these compounds can also be reduced to a minimum before the SCR - stage.

Evaluating the requirements here of:

- the very best collector for submicron particles
- a device reducing the acidic gases SO_2, SO_3 and HCl as much as possible,

the answer is dry scrubbing with fabric filter.

This conclusion is drawn from many comparisions of flue gas cleaning systems for solid waste incinerators. One of the first evaluations was published in No 4 of Waste Management and research 1986 (ref 5). This evaluation can be completed with some new measurements - table 1

TABLE 1 EMISSION LEVELS IN MICROGRAM/M^3 NORM DRY GAS
(FOR MERCURY: TOTAL EMISSION GASEOUS + PARTICULATE)

APC - configuration	Dust	Zinc	Lead	Cadmium	Mercury	Ref.
Dry scrubber+FF	1 000- 8 000	22	10	0.09	5-50	Ref. 6
Dry scrubber+FF	1 000		15	0.3	5	Ref. 7
ESP+Wet scrubber	500- 6 000	30- 350	10- 200	1- 10	5- 10	Ref. 8
ESP+Wet scrubber	1 000- 5 000	1 720	130	12	6	Ref. 9
ESP+Wet scrubber with condensation	4 200	356	144	6.5	47	Ref. 10
Dry scrubber+ESP	<5 000	Pb + Zn = 500		10		Ref.11
Spray dryer+cyclone +wet scrubber	15 000- 30 000	1 350- 6 590	430- 1 050	34- 120		Ref. 12
ESP+wet scrubber	1 000- 8 100	110- 560	50- 180	10- 24	32- 34	Ref. 13
ESP+wet scrubber	1 000	60- 200	150- 160	15- 25	12- 50	- " -
ESP+wet scrubber	300- 5 000	100	40	33	20	- " -

We also see here very evidently the tendency - if we want to minimize the concentration of heavy metals we have to go for a fabric filter.

Dioxins are also removed very efficiently in this dry scrubber/FF combination (ref. 13).

APC - SYSTEM FOR MUNICH SOUTH

In the inquiry for the APC - system for Munich South the following emission limits were required (table 2).

TABLE 2 EMISSION LIMITS FOR THE MUNICH SOUTH INCINERATOR PLANT

	Emission mg/m^3 n.t.p.	ppmV	Removal efficiency percent
Particulates	10		> 99.9
HCl	10	6	> 99
SO_2	25	9	> 95
NO_x	70 (as NO_2)	35	> 88

Fläkt tendered for and received the order with a CDAS dry scrubber/fabric filter combination followed by a SCR $DeNO_x$ - system operating at approx. 320 °C (610 °F). Figure 1 shows the principle of the APC - plant for one unit. Munich South has two units each with a capacity of about 1 000 ton waste/day.

To obtain this high removal efficiency at low levels of excess lime the dry scrubber has to be operated at a low temp or in this case approx. 125 °C (255 °F).

The first experiences with low temperature operation and low HCl/SO_2 and dust emissions were obtained at the pilot in Quebec City operated jointly by Environment Canada and Fläkt (ref 14).

The units 1 and 2 at the municipal solid waste incinerator in Stockholm - the Högdalen plant - were started in October 88 and have since been in continuous operation at this low temperature.

CHOICE OF CATALYST FOR MUNICH SOUTH

The SCR DeNO$_x$ - stage location downstream of the dry scrubber/fabric filter combination - a "tail end" location - makes it possible to choose a very effective heat exchanger and catalyst.

Subsequently all makers of catalysts were very interested in quoting for this plant. However, as the experiences of this application were very limited, it was decided to test the most promising catalysts in a pilot plant located downstream of a dry scrubber/FF installation at MSW-incinerator unit 3 Kempten Bavaria. The unit incinerates approx. 7 ton MSW/h in a conventional stoker furnace followed by a steam boiler down to approx. 200 °C (390 °F), where the CDAS dry scrubber/FF is connected.

PILOT PLANT AT MSW - PLANT KEMPTEN BAVARIA

The DeNO$_x$-pilot was operated according to the tail end concept; downstream of a CDAS dry scrubber followed by a fabric filter. The catalytic reactor was divided into three parallel lines, each with a catalyst capacity of 0.2 m^3, see figure 2. Through each reactor line the flue gas flow could be varied up to 1 500 Nm^3/h.

The flue gas firstly passed through a non-leakage tubular contra-flow heat exchanger where the gas temperature was increased from 140 °C to 275 °C (285 °F to 530 °F). After an electric heater the temperature finally reached the reaction temperature 320 °C (610 °F).

Ammonia vapour was injected into the gas duct through a nozzle arrangement shortly after the electric heater, and a static mixer ensured good mixing of the ammonia with the flue gas.

Ammonia was evaporated directly from pressure bottles at a temperature of 10-15 °C (50-60 °F) by using a water bath with circulating water. Before injecting the ammonia vapour into the gas duct the pressure was reduced and the ammonia flow was measured by a mass flow meter.

The cleaned gas from the three reactors merged in to one gas duct, passed through a venturi, where the total gas flow was measured, and the heat exchanger before it was returned to the existing stack. The outlet temperature was approx. 180 °C (355 °F).

The gas flow through each reactor line was adjusted by using the hand-operated damper at the raw gas side and the three valves installed at the reactor outlet.

Each reactor was equipped with at least two, to maximum four, catalyst stages and with two catalyst casettes containing small pieces of catalyst to be used for activity measurements in laboratory. A catalyst stage took 0.05 m^3 of catalyst.

The pilot had good thermal insulation to avoid heat losses and temperature gradients. The reactor was also equipped with electric heating elements in order to compensate for small heat losses. The pilot was well sealed to prevent leakages.

ANALYSER SYSTEM

Continiously working analysers for water vapour, nitrogen oxide, nitrogen dioxide, sulphur dioxide and oxygen were installed in the pilot together with a mass flow meter for the injected ammonia (figure 2). The content of sulphurtrioxide and unreacted ammonia were both analysed from gas samples.

Analysers used:

Component (s)	Analyser
NO	Maihak, UNOR 6N (IR) (two instruments)
NO_2, SO_2	Leybold- Heraeus, Binos (UV)
H_2O	Bodenseewerk, Spectran 677 (IR)
O_2	Westinghouse, zirconium- oxide sensor
NH_3	Brooks, mass flow meter

Sample gas for analysis was tapped from the rawgas side in a point just upstream the ammonia injection, and from the clean gas side in points after each catalyst stage (figure 2). The gas first passed a measuring probe equipped with a particulate filter and was then transported in heated and insulated (90 °C) (195 °F) teflontubes, to a refrigerator where it was dried to a dewpoint of + 2 °C (35 °F). After the dryer a diaphragm pump transported the gas, still in heated teflon tubes, to the analysers in the control room.

Water vapour was measured on the raw gas side directly from hot flue gas (180 °C, 355 °F) and the analyser was fixed to this measuring point. Two oxygen sensors were used. One was installed upstream of the ammonia injection and the other just downstream of the venturi on the clean gas side.

The instruments measuring NO, NO_2 and SO_2 could be connected to any of the two test gas lines and more than one analyser could be used at the same measuring point. At the reactors, a gas sampling point was chosen by manually moving the test gas tube between the probes. This was easily done by using quick-connection joints between the pipe and probes. One of the NO-analysers was normally fixed on the raw gas side, used also for the ammonia controlling.

In parallel to this conventional measuring system, tests with an in-situ analyser at two measuring points have also been made. The analyser worked following the DOAS-principle (Differential Optical Absorption Spectroscopy), and was manufactured by the company OPSIS AB in Sweden. The instrument measured NO, NO_2, SO_2 and NH_3.

The measuring methods used for ammonia and sulphurtrioxide have been developed by the Electrical Power Development Co LTD (EPDC) in Japan. The ammonia method is based on absorption in an acidic water solution and the sulphurtrioxide method on the controlled condensation method.

CONTROL SYSTEM

The pilot plant was run automatically and controlled by a process computer. The reactor temperature could be set between 140 °C and 400 °C (285-750 °F) and controlled by temperature measurements on the reactor inlet, by means of the electric heater.

When the catalyst performance was studied, the ammonia flow was normally controlled in order to keep a constant mole ratio (NH_3/NO_x) at the reactor inlet. From the flue gas flow and the concentration of nitrogen oxide and water vapour the actual nitrogen oxide flow was calculated. Together with the chosen mole ratio (NH_3/NO_x), this gave the ammonia flow to be injected to the plant.

For the full scale plant in Munich, the guarantees are based on a constant NO_x outlet level combined with a maximum slip of ammonia. This "constant outlet NO_x" controlling has also been tested in the pilot in order to follow both load swings and variation in NO_x-flow at constant load. The control loop used was a feed forward control on the mole ratio with backtrim from measurements on the actual outlet NO_x-concentration.

TEST PROGRAM

The catalyst performance was measured alternatively with fixed and variable settings of flue gas flow and mole ratio (NH_3/NO_x). The reaction temperature was kept on 320 °C (610 °F).

Fixed settings per reactor line:

- Flue gas flow (Nm^3/h)	1 000
- Reaction temperature (°C)	320
- Mole ratio (-)	0.9

For each setting of these variables the ammonia slip was measured manually after the different catalyst levels.

To judge changes in catalytic activity with operating time, catalyst samples have been analysed in a laboratory reactor under constant conditions.

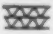
TECHNICAL DATA

- Reactors in parallel		3
- Nominal gas flow to each reactor	Nm^3/h	1 000
- Nominal reaction temperature	°C	320
- Oxygen concentration	% vol	10
- NO_x inlet concentration	ppm	200-250
- HCl inlet concentration	ppm about	30
- SO_2 inlet concentration	ppm about	30
- SO_3 inlet concentration	ppm	< 1
- Dust inlet concentration	mg/Nm^3	< 5
- NO_x-reduction	%	90

CATALYSTS

All tested catalysts were of the parallel flow type and both the honeycomb and the plate type catalysts were tested in the pilot. The active materials were based on V_2O_5/TiO_2 in all cases (see table 3).

TABLE 3 CATALYST SPECIFICATION

Catalyst	A	B	C
Active material	V_2O_5-TiO_2	V_2O_5-TiO_2	V_2O_5-TiO_5
Catalyst type	honeycomb	plate	honeycomb
Configuration			
Pitch (mm)	4	3	4
Specific area (m^2/m^3)	750	1 000	750
Number of used catalyst stages	3	2	4

The catalytic reduction of NO and NO_2 in the presence of oxygen proceeds in the following way (ref 16):

(1) $4\ NO + 4\ NH_3 + O_2 \quad 4N_2 + 6\ H_2O$

(2) $2\ NO_2 + 4\ NH_3 + O_2 \quad 3N_2 + 6\ H_2O$

Since usually at least 90 % of the NO_x is in the form of NO, the reaction (1) will become the most important.

The two most important characteristics of a $DeNO_x$-catalyst are the specific surface and the chemical activity. The specific area is defined as the macroscopic area divided with the total catalyst volume.

The catalytic reaction is a pseudo first order reaction with respect to the NO-concentration which can be expressed as (ref 15):

$$-AV \cdot \ln(1-x) = k$$

where; $AV = \frac{\text{Flue gas flow } (Nm^3/h)}{\text{Catalyst surface area } (m^2)}$ = area velocity (Nm/h)

k = catalyst activity constant (Nm/h)

x = NO_x-reduction (-)

The constant k, usually called the activity constant, is a measure of the catalytic activity of a certain catalyst. This constant is temperature dependent and of course also dependent on the composition of the active material which is illustrated in figure 3.

When increasing the amount of active material and/or increasing the reaction temperatures a rapid increase in SO_2 conversion to SO_3 will result. In a commercial catalyst, used on SO_2 containing flue gases, the SO_2 conversion is kept around 0.5-1 %.

The catalysts are principally of the plate - or the "honeycomb" - type due to the used manufacturing method. In the extruded honeycomb catalyst the walls are self-supporting and built only of the active material. The material is brittle and therefore shock sensitive. The plate type has a built in matrix of metal which gives thinner catalyst walls and better shock resistance.

CATALYST DEACTIVATION

As in all processes using heterogeneous catalysts the catalytic activity decreases with increased operating time due to both chemical and physical factors:

- Thermal sintering, decrease in the microscopic specific surface.
- Erosion and accumulation of dust in fine catalyst pores.
- Pore condensation of different salts such as ammoniumsulphates, ammoniumcloride, arsenic oxide, zinc chloride etc depending on temperature and flue gas composition.
- Chemical poisoning from alkali metals, mercury, arsenic, selenium etc.

It has been found (ref 16) in Japan that erosion and dust effects dominate the deactivation process in coal fired dry bottom boilers.

The effects of dust are minimized in the tail and process and when connected to a waste incinerator the most interesting items would be the pore condensation and chemical poisoning, because of the large number of metal salts which can occur in these flue gases. The experience from waste incinerator flue gas is rather limited.

The relative activity, defined as the activity constant after t hours of operation (k_t) relative to the original constant (k_o), is used as a measure of the catalytic loss (ref 16). Depending on which deactivating process dominates, the deactivation curve will have a different slope.

In practice, the deactivation can only be compensated for by an increased mole ratio, but only to a point where the NH_3-slip guarantee still can be kept, or by increasing the reaction temperature.

RESULTS AND OPERATING EXPERIENCES

All tested catalysts responded with a pseudo first order reaction behaviour when the ammonia amount was increased up to a certain mole ratio. Beyond this mole ratio the NO_x-reduction became a fixed value. The point where this pseudo first order kinetics terminated was dependent on the space velocity (SV); the gas flow per volume of catalyst.

In figure 4, response curves for the three catalysts are shown. All measurements were made at 320 °C (610 °F) for different space velocities. Catalyst B showed an almost theoretical dependency between the mole ratio and NO_x-conversion at least up to 20 000 1/h in space velocity (SV). The two honeycomb catalysts showed a deviation from this curve at least at SV-values above 18 000 1/h. When comparing this behaviour one must also take into account that the honeycomb catalysts had only 75 % of the plate catalysts specific area.

At low SV-values, all catalysts consumed ammonia to reduce nitrogen oxide in a ratio very near 1:1. This "theoretical" behaviour can also be seen in figure 5 where space velocity vs. ammonia slip is shown at 320 °C (610 °F). All curves represent a mole ratio of 0.9 +/- 0.05. At low SV-values the ammonia leakage was very small but at a certain value the slip increased very rapidly also indicating a decrease in NO_x-conversion from the theoretical value. This behaviour is shown for the two honeycomb catalysts A and C. The breakpoint was in the SV-range 12 000 - 18 000 1/h. The plate catalyst operated on the theoretical line even up to SV = 20 000 1/h.

Both figures represent measuring data over a period of 2 300 h of operation. From this data, no change in catalytic activity could be seen.

As the main reason for this pilot testing was to check the catalytic activity in waste incinerator flue gases, activity measurements were also made in lab reactors by the manufacturer (catalyst A and C) and by the University of Karlsruhe, "Institut für Chemische Technik" (catalyst B). After 2 300 hours of operation, the measurements showed a deviation in catalytic activity in the range of +/- 1 % from the original activity.

Measurement for N_2O and dioxins over the reactor was also made. The N_2O analyses was made by a research institute (IVL) in Gothenburg, Sweden. Gas samples were taken both from the raw gas and clean gas side and analysed with GC-technique. No N_2O was found in these samples. The dioxin measurements were made by (Energi Miljö Konsult AB) EMK, Malmö, Sweden, using the University of Umeå, Prof. Rappe for the analysis. The concentration of dioxins was very low both at the raw gas and clean gas side, see table 4. No dioxins were produced over the catalyst.

TABLE 4 DIOXIN CONCENTRATIONS CALCULATED IN DIOXIN EQUIVALENTS ACC TO EADON AT SCR - $DeNO_x$ - PILOT PLANT KEMPTEN BAVARIA.

Catalyst temperature	320 °C (610 °F)	280 °C (540 °F)
Outlet from fabric filter ng/m^3n dry 10 % CO_2	< 0.05	< 0.01
Outlet from SCR $DeNO_x$ ng/m^3n dry 10 % CO_2	< 0.01	< 0.01
NO_x - emission ppm	30	30
NO_x - reduction (%)	90	90

There was no dust build up in the reactor or in the heat exchanger after 2 300 hours of operation and the pressure drop over the reactor has also remained constant over the period.

CONCLUSION

The operating experiences with the pilot plant were extremly good and the results, especially from the catalytic activity measurements, were also an important factor when Stadtwerke München Süd gave Fläkt our first order on a $DeNO_x$-installation at a waste incinerator plant. This will be the first SCR-plant connected to a waste incinerator in West Germany.

FURTHER DEVELOPMENT

After receiving the order from Stadtwerke München Süd, new catalysts were run in the pilot at 250 °C (480 °F). This temperature is probably more critical, compared to 320 °C (610 °F), for pore condensation and metal salt accumulation on the catalyst. Activity measurements have not yet been made on these catalysts.

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Fläkt

APC - for MSW - Incinerator Munich South

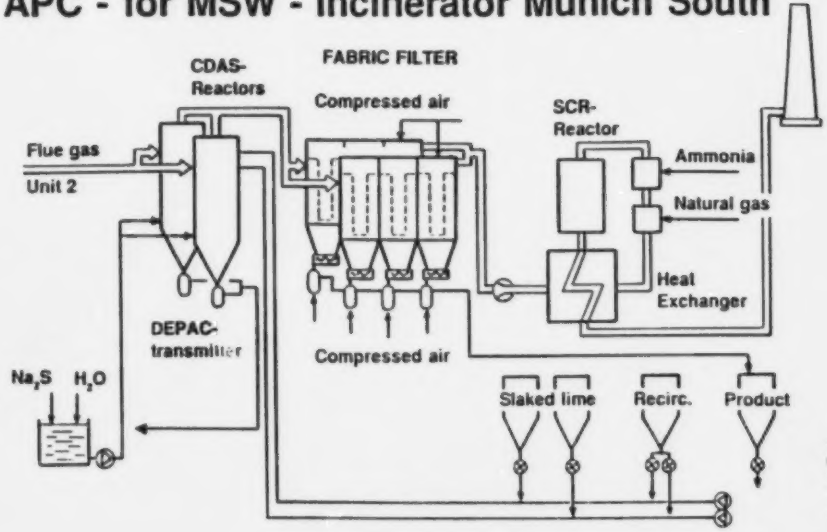

CDAS-
Reactors
FABRIC FILTER
Compressed air
SCR-
Reactor
Flue gas
Unit 2
Ammonia
Natural gas
Heat
Exchanger
DEPAC-
transmitter
Compressed air
Na₂S
H₂O
Slaked lime
Recirc.
Product

Figure 1.

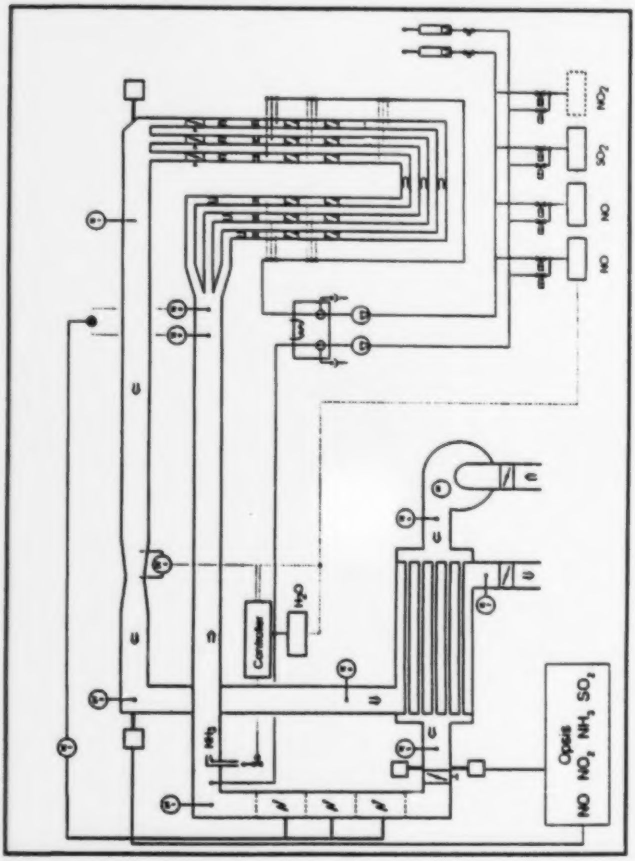
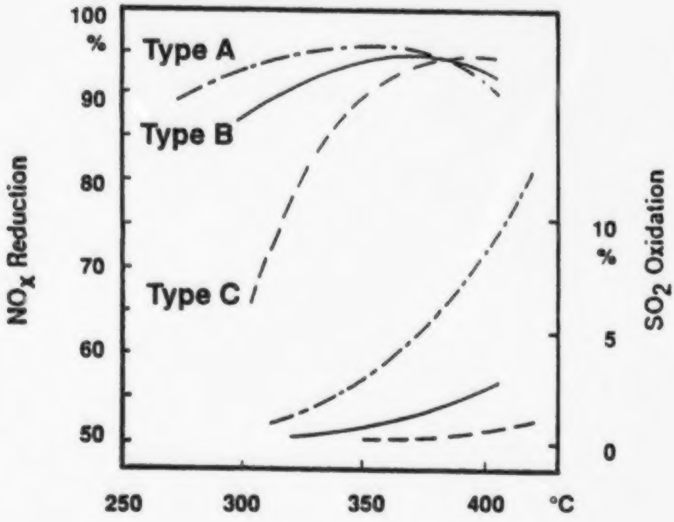

Opsis
NO, NO₂, NH₃, SO₂
NO
NO
SO₂
NO₂
Controller
H₂O
NH₃

Figure 2. Flow sheet, SCR-pilot plant at the MWC plant in Kempten FRG.



Type	Active material	Field of application
A	High	Clean gas
B	Average	Clean gas
C	Low	Raw gas

Figure 3.[1] *Catalytic NO_x reduction and SO_2-oxidation by temperature and amount of active material.*

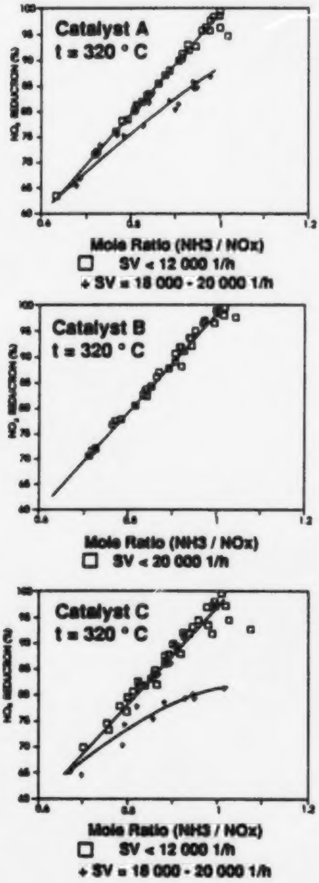


Figure 4. Mole Ratio (NH3/NOx) vs. NOx-reduction at different space velocities (SV).

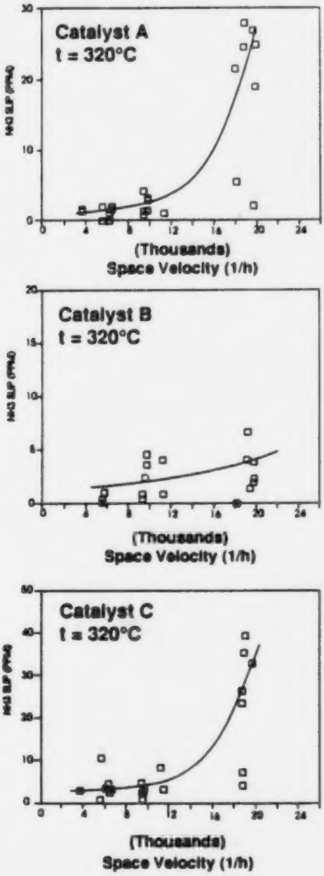


Figure 5. Space velocity vs. unreacted ammonia at mole ratio 0.9 ± 0.05.

CHARACTERIZATION OF A LIME SPRAY DRYER

ABSORBER/BAGHOUSE ON A REFUSE-DERIVED FUEL COMBUSTOR

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ABSTRACT

The U. S. Environmental Protection Agency and Environment Canada are jointly sponsoring characterization and performance testing on Unit 11 of the Connecticut Resource Recovery Authority's Mid-Connecticut Facility in Hartford. The characterization testing (Phase 1) was completed on January 24, 1989, while performance testing (Phase 2) began on February 13, 1989. The approach to the overall test program is briefly discussed. Characterization test results for the flue gas cleaning (spray dryer absorber/baghouse) system are presented and discussed. These results include the combustion test conditions, inlet acid gas concentrations, and acid gas control. Where possible, the results are compared with those obtained during the July 1988 state compliance tests.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and cleared for presentation and publication.

INTRODUCTION

Both the U.S. Environmental Protection Agency (EPA) and Environment Canada (EC) have performed field tests on municipal waste combustors (MWCs) and their flue gas cleaning (FGC) systems in their evaluations of various combustion and air pollution control system designs. With the exception of the Marion County, OR, tests (1) and the recently completed Dayton, OH, tests, the EPA effort focused on pollutant data collection during state-required compliance tests. These tests were made in cooperation with the host MWC plants in Biddeford, ME (2), Millbury, MA (3), and Hartford, CT (4). Under its National Incinerator Test and Evaluation Program (NITEP), EC evaluated mass burn [Quebec City (5)] and two-staged combustion [Prince Edward Island (6)] systems. For the mass burn plant in Quebec City, one of the furnaces was redesigned with state-of-the-art features. In addition, pilot-scale air pollution control systems using scrubbers/fabric filters were evaluated (7). These modifications improved combustion and significantly reduced organic emissions. Both EPA and EC have also collected other data on MWC performance in their separate regulatory development and support programs.

A joint EPA/EC program to evaluate the combustion and air pollution control systems for a modern refuse-derived fuel (RDF) unit is currently in progress and is designed to evaluate these systems for both normal operation and potential upsets. The test program, to be discussed in more detail below, is being conducted in two phases on Unit 11 at the Connecticut Resource Recovery Authority's Mid-Connecticut facility in Hartford. Constructed and operated by Combustion Engineering, Inc. (CE), this facility has combustion and lime spray dryer absorber/baghouse systems which are representative of state-of-the art designs.

While the two-phased test program evaluated the effects of combustion variables on pollutant emissions, such as acid gases, organics, trace metals, and particulate matter, and FGC system variables controlling these emissions, only the results of the Phase 1 (characterization tests) will be discussed here, as the Phase 2 (performance tests) was in progress during the preparation of this paper. This paper will also focus on the FGC system since another paper being presented at this conference discusses the combustion system evaluation. In addition to addressing the results for the FGC system during the current program, a comparison with the results obtained during the July 1988 compliance tests (8) will be made where possible. It is also noted that the short time between the completion of the characterization tests and the preparation of this paper precluded a complete evaluation and discussion of the test results.

TEST PROGRAM

As noted above, the test program for Unit 11 at the Mid-Connecticut facility consists of two phases: characterization and performance. The 28 characterization tests were conducted during January 11-24, 1989, several months later than originally planned due to the unavailability of the unit to be tested. These tests were short-term ones, normally lasting about 2 hours after combustion conditions were set and stabilized. Two tests involving organics sampling lasted 6 hours (1-hour sampling time in each of four ports). Several tests under poor combustion conditions were ended prematurely after only an hour or less.

Combustion variables included boiler steam load, the amount and distribution of tangential overfire air, the amount of rear wall overfire air, the forced draft fan supply pressure, undergrate air distribution, and ratio of overfire to underfire air. The controlled variables for the flue gas cleaning system were the gas temperature leaving the spray dryer absorber and the SO_2 concentration at the fabric filter (baghouse) outlet. With these quantities as set points and other measured process data, the approach to saturation temperature and stoichiometric (reagent) ratio (the major variables governing spray dryer absorber performance) can be calculated for varying flue gas temperatures and acid gas concentrations at the inlet to the spray dryer absorber.

The performance tests are designed to determine the emissions of acid gases, selected organics and trace metals, and particulate matter and their control under test conditions determined from the study of the characterization test results. Fourteen tests are planned, and they will include triplicate runs for selected conditions. Process quantities measured during the characterization phase (see Test Instrumentation and Methods, below) will be supplemented by RDF sampling, residue sampling from various locations, and sampling to determine organic, trace metal, and particulate matter emissions and their control, along with data on acid gases, across the spray dryer absorber/baghouse during the performance tests. During several of the performance tests, organic sampling of the flue gas stream is planned at the inlet to the air heater simultaneously with organic sampling at the spray dryer inlet to determine if dioxins/furans are formed in the air heater where the gas temperature ranges from about 700 to 400°F.* Sampling at the baghouse outlet will also be performed in several tests to determine the size distribution of the particulate matter and concentration of hexavalent chromium in the cleaned flue gas.

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*English Engineering units rather than metric units are generally used here. See the Appendix for unit conversion factors.

DESCRIPTION OF UNIT TESTED

Unit 11 (see Figure 1) is one of three units of identical design at the Mid-Connecticut facility. The unit is rated at 231,000 lb steam/hr at 880 psig and 825°F when firing 100% RDF at a nominal rate of 675 tons/day. Each unit can fire coal only or a combination of coal and RDF. During the tests reported here, the unit burned RDF only in the CE spreader-stoker boiler rated at 675 tons RDF/day.

Combustion air is normally supplied to the furnace by a single forced draft fan through 10 undergrate damper-controlled air plenums, 5 plenums in two parallel rows under the grate moving from the rear to the front (RDF feed end) of the furnace, and overfire ports located at the corners of the furnace at three vertical levels. The three vertically aligned overfire ports in each corner contain nozzles which can be directed horizontally to either side of the diagonal plane passing through the opposite corners of the furnace in order to promote the mixing of air with combustion gases. The nozzles at each corner are fed by a common plenum. The horizontal orientation of each nozzle can be individually controlled, but the normal practice is to have the same orientation for all four nozzles at each level (bottom, middle, or top). Overfire air can also be supplied through the rear wall of the furnace using a separate forced draft fan to boost the combustion air pressure for air provided from the rear wall.

RDF is supplied by the municipal solid waste processing plant located adjacent to the MWC plant. RDF is pneumatically injected into each boiler from four distributors located on the front furnace wall above the travelling grate. The grate speed and RDF feed rates are adjusted to control the depth of the bed on the grate to accommodate the desired steam load. The steam generated in the MWC system is sold to the Connecticut Light and Power Company and is converted to electrical energy in an on-site turbogenerator operated by CE.

Unit 11 is equipped with a lime spray dryer absorber having a single rotary atomizer which injects fine slurry droplets into dirty flue gas distributed concentrically around the droplets from the top of the spray chamber to effect good gas/droplet contact. The absorber is designed to handle about 165,000 cfm of flue gas at 340°F and to remove 90% HCl and 70% SO_2 at inlet concentrations of approximately 700 and 150 ppm for HCl and SO_2, respectively. The cooled gas with entrained particulate matter, reaction products, and unreacted lime is then ducted through a heating section, which supplies heat only when needed to preclude a gas temperature of 225°F or less, to the baghouse.

Lime slurry to the atomizer is pumped from the feed tank serving the unit. This feed tank receives slaked lime slurry from the storage tank serving all three of the plant's MWC units and dilution water from the coal pile drainage/ash/residue pond or, when this supply is insufficient, from the Connecticut River. Thus, the solids [$Ca(OH)_2$] content of the atomizer feed can be controlled to effect the desired acid gas removal, and at the same time provide sufficient water to control the flue gas temperature at the spray dryer outlet.

The baghouse has 12 compartments and uses reverse air cleaning, which can be activated manually or by a set pressure drop across the bags in each compartment. Each compartment contains 168 Teflon-coated glass-fiber bags. Each bag is 8 in. in diameter and 24.5 ft long. The baghouse has a gross air-to-cloth ratio of 1.45 cfm/ft^2 and a net air-to-cloth ratio of 1.74 cfm/ft^2 when two compartments are off-line (one for cleaning and one for maintenance).

Residues from the grate (bottom ash plus siftings), economizer, air heater, and baghouse are combined prior to disposal. They are trucked, along with other non-combustibles, to a landfill for disposal.

TEST INSTRUMENTATION AND METHODS

Continuous emission monitors (CEMs) were used to measure gaseous CO, O_2, CO_2, HCl, SO_2, NO_x, and total hydrocarbons. These monitors were calibrated using EPA- and EC-approved methods. Manual methods were also used to determine the moisture, CO_2, O_2, and HCl content in the flue gas. Moisture content in the gas entering the spray dryer absorber was also determined with a non-dispersive infrared monitor. Gas flow rates at the sampling train locations were determined using pitot tube traverses. The steam flow, air flows, atomizer slurry flow, lime slurry flow, and most pressure and temperature data were obtained from plant instrumentation. Other pressure and temperature data were measured using equipment installed for the tests. A data acquisition system acquired real time data and was used to process them into the desired form. The sampling trains used were checked and used in accordance with EPA- and EC-approved protocols.

Figure 1 shows the locations of the CEMs and sampling ports used in the characterization tests.

DISCUSSION OF CHARACTERIZATION TEST RESULTS

Table 1 summarizes the stabilized characterization test conditions for both the combustion and FGC systems. The establishment of a given set of combustion conditions was determined by observing the steam load, excess O_2, and CO over several hours. When these quantities were stabilized, the test was begun, with the CO value measured at the spray dryer absorber being the main criterion for good or poor combustion. The upper three curves of Figure 2 are examples of the steam flow, excess O_2, and CO monitored during a characterization test after the combustion had stabilized. A more detailed discussion of the combustion component of the characterization tests will be provided in another paper at this conference and thus will not be covered further here.

Because the unit tested fires RDF which has variable composition and properties and is required to meet acid gas emission limits, the normal control parameters for spray dryer absorber performance [approach to saturation temperature at the absorber outlet and stoichiometric (reagent) ratio] were replaced by gas temperature at the spray dryer outlet and SO_2 concentration at the fabric filter (baghouse) outlet. As noted earlier, these quantities plus other measured data can be used to obtain the approach to saturation temperature and stoichiometric ratio.

The results for the flue gas cleaning system tests are portrayed in Figures 2-4 and Tables 2-5. Figure 2 (lower six curves) depicts quantities measured as a function of time which are characteristic of the flue gas cleaning system. As noted earlier, the controlled parameters were the gas temperature at the spray dryer absorber outlet (SDA-O-Temp) and the SO_2 concentration in the cleaned flue gas at the fabric filter outlet (FA-Out-SO_2). These curves show that these parameters were well-controlled (i.e., they were almost constant and equal to their set-point values), while the inlet SO_2 concentration of the gas entering the spray dryer absorber (Sp-In-SO_2) ranged from about 110 to 240 ppm over the 1-hour period, with the average value approximating 170 ppm (SO_2 in dry gas, but not referenced to 12% CO_2).

Figure 3 gives the average HCl concentrations (dry gas, referenced to 12% CO_2) across the flue gas cleaning system as a function of the gas temperature at the outlet of the spray dryer absorber. In addition, the HCl removal in the spray dryer absorber/baghouse system is shown to vary only slightly with this temperature, the removal being over 90% for 24 of the 27 characterization test runs. (Data for the first run were not available for computing the HCl removal.) Although the average inlet HCl concentrations varied by a factor of two and were all above 300 ppm, the average outlet HCl concentrations never exceeded 75 ppm for even the minimum lime slurry flow. Also shown in Figure 3 (solid symbols) are the HCl concentrations and HCl removal from the July 1988 compliance tests (8). For these three runs, the average HCl removal was over 98%, while the outlet HCl concentration ranged from 3 to 5 ppm.

The results shown in Figure 4 for SO_2 are much more scattered than those for HCl, and might be expected since the HCl is more reactive with the $Ca(OH)_2$ sorbent used than is SO_2. As planned for the characterization test phase, the SO_2 removal ranged from a high of 100% (Run 11C) to essentially zero. Several runs indicated higher SO_2 outlet concentrations at the baghouse than were measured at the spray dryer absorber inlet (see Table 2; Runs 8, 9, 10A, and 10B). Except for Run 9, no lime slurry was fed to Unit 11 during these short-term tests. Since the outlet SO_2 concentrations were 20% or more than those at the spray dryer absorber inlet and the values were obtained using calibrated SO_2 monitors, the values are believed to be correct. These outlet values, which exceeded the inlet concentrations, may have resulted from SO_2 being evolved from sulfite ions in the dilution water. The dilution water from the coal pile/ash/residue drainage pond is supplied to the atomizer slurry feed tank and later atomized in the spray dryer absorber where the flue gas temperature ranges between 300 and 400°F. Thus, the sulfite ions in the presence of hydrogen ions and heat from the flue gas may have led to the formation of SO_2 in the spray dryer absorber in the absence of $Ca(OH)_2$ in the atomizer feed slurry.

The SO_2 removal for all spray dryer absorber gas outlet temperatures (240-340°F) was greater than 70% when the atomizer slurry feed contained sufficient $Ca(OH)_2$ solids and was supplied at a rate adequate to maintain a 50-ppm or lower SO_2 concentration at the fabric filter outlet.

For the July 1988 compliance tests, the average SO_2 emission was reported to be 0.002 lb $SO_2/10^6$ Btu, ranging from 0 to 0.004 lb $SO_2/10^6$ Btu for the three runs. Using the permitted emission limit of 0.32 lb $SO_2/10^6$ Btu as being equivalent to 120 ppm for this unit, the SO_2 concentration in the flue gas at the fabric filter outlet would have ranged from 0 to 1.5 ppm (0.75 ppm average) during the compliance tests. Since the inlet concentration SO_2 values were not reported, SO_2 compliance test data are not shown in Figure 4.

SUMMARY

Characterization tests, the Phase 1 of a two-phased test program, on Unit 11 of the Mid-Connecticut Resource Recovery Facility in Hartford, have been completed. The lime spray dryer absorber/fabric filter system was characterized for a range of spray dryer outlet gas temperatures and SO_2 concentrations in the clean flue gas at the fabric filter outlet when the RDF-fired combustor operated under both normal and potential upset conditions. The flue gas cleaning system showed preferential removal of HCl over SO_2 as expected. The HCl removal ranged from 82 to 98% when the gas temperature at the spray dryer outlet and SO_2 concentration in the flue gas at the fabric outlet were the controlled variables. The HCl removals reported for the July 1988 compliance tests were 98-99%, and are comparable to values obtained with high lime slurry flows during the characterization tests. The SO_2 removal for the characterization tests ranged from 0 to 100%, with the lowest value being obtained for several tests with no lime slurry flow. In these tests, the SO_2 concentration in the gas at the fabric filter outlet exceeded the SO_2 concentration at the spray dryer inlet. This unexpected result may have been caused by sulfite ions present in the water contained in the atomizer slurry feed being converted to SO_2 in the spray dryer absorber at the prevailing temperatures when no $Ca(OH)_2$ was in the feed slurry. The flue gas cleaning system was capable of maintaining high removals of HCl and SO_2 for inlet concentrations varying by a factor of about two for both of these acid gases.

ACKNOWLEDGMENTS

The cooperation of the Connecticut Resource Recovery Authority, Combustion Engineering, Inc., and Alliance Technologies, Inc. (the test program contractor for EC) in the characterization tests is sincerely appreciated. The assistance of R. Michael Hartman, coordinator for Combustion Engineering during the EPA/EC test program, is gratefully acknowledged.

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APPENDIX

Unit Conversion Factors

English Engineering Unit	Multiplied By	Metric Unit
in.	2.540×10^{-2}	m
ft	3.048×10^{-1}	m
ft^2	9.290×10^{-2}	m
ft^3/min. (cfm)	4.719×10^{-4}	m^3/s
gal./min. (gpm)	6.312×10^{-5}	m^3/s
klb/hr*	4.536×10^{2}	kg/hr
lb/hr	4.536×10^{-1}	kg/hr
lb/10^6 Btu	4.302×10^{-7}	kg/kJ
psi	6.895×10^{3}	Pa
in. water	2.490×10^{2}	Pa
ton/day	9.070×10^{-1}	tonne/day
°F-32	5.555×10^{-1}	°C

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*Note that KPH in Figure 2 represents klb/hr.

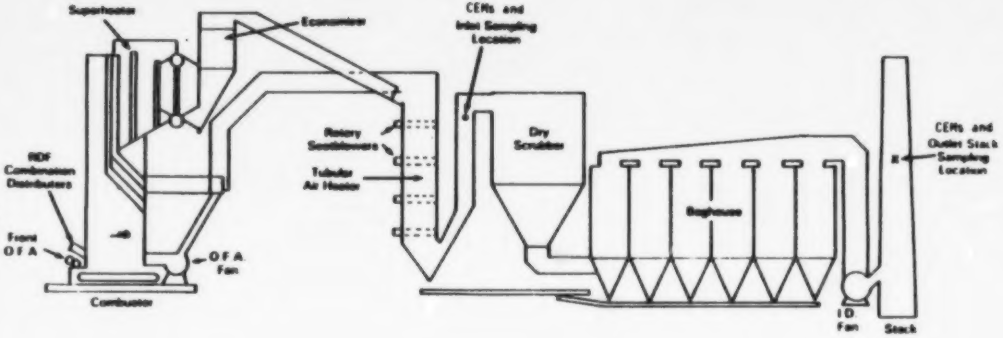

Superheater
Economizer
CEMs and
Inlet Sampling
Location
RDF
Combustion
Distributors
Front
O.F.A.
O.F.A.
Fan
Combustor
Rotary
Sootblowers
Tubular
Air Heater
Dry
Scrubber
Baghouse
CEMs and
Outlet Stack
Sampling
Location
I.D.
Fan
Stack

Figure 1. Schematic flow diagram for Unit 11 of the Mid-Connecticut Resource Recovery Facility with sampling locations.

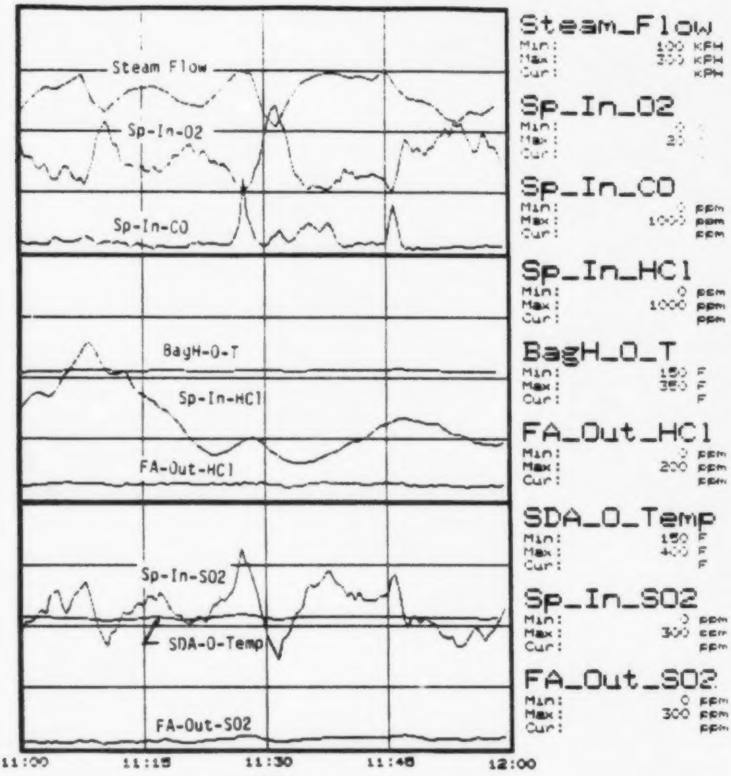


Figure 2. Characterization test data: process and continuous emission monitor data (SP=SDA=spray dryer absorber; In=inlet; BagH=FA=fabric filter; O=Out=outlet; T=Temp=temperature; KPH=10^3 lb/hr).

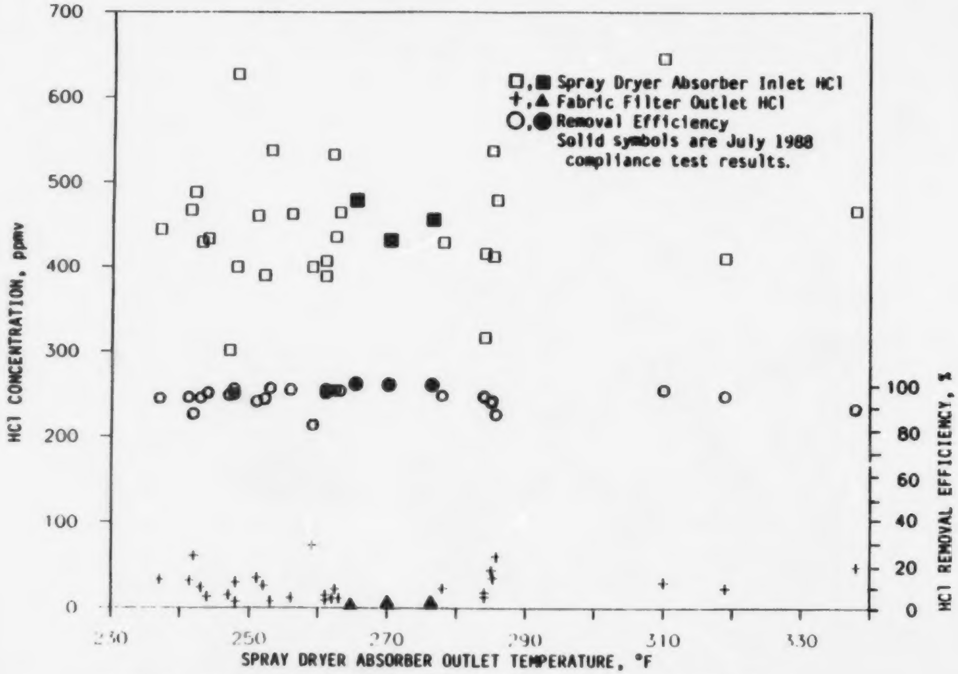


Figure 3. Characterization test data: HCl concentration and HCl removal efficiency for flue gas cleaning system (concentrations referenced to 12% CO_2).

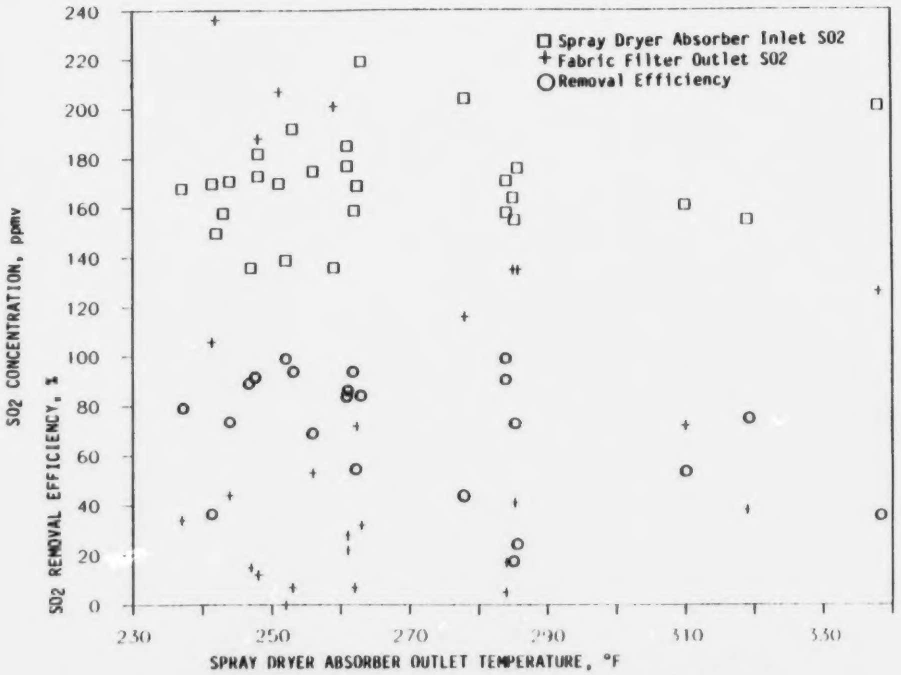


Figure 4. Characterization test data: SO_2 concentration and SO_2 removal efficiency for flue gas cleaning system (concentrations referenced to 12% CO_2).

TABLE 1. CHARACTERIZATION TEST CONDITIONS

Date	Test No.	Nominal Steam Load[a] 10^3 lb/hr	Levels of Tangential Overfire Air	Underfire & Tangential Overfire Air Fan Pressure in. Water	Rear Wall Overfire Air Fan Pressure in. Water	Average Spray Dryer Absorber Outlet Temperature °F	Average Fabric Filter Outlet SO_2 Concentration[b] ppmv	Test Duration min.
1/11/89	CT01	231	3	15	0	NA[c]	131	45
1/12/89	CT02	231	3	15	0	206	135	120
1/12/89	CT03	190	3	15	0	241	106	120
1/12/89	CT04	231	3	10	35	285	41	120
1/13/89	CT05	150	1	15	0	244	44	120
1/16/89	CT06	231	2	15	0	262	72	120
1/16/89	CT07	231	3	15	0	285	135	120
1/16/89	CT08	231	2	15	35	251	207	120
1/17/89	CT09	231	2	15	15	248	188	120
1/17/89	CT10A	150	3	15	0	242	236	75
1/17/89	CT10B	190	3	15	0	259	201	45
1/17/89	CT11A	150	0	15	0	237	34	60
1/17/89	CT11B	150	0	15	15	243	NA[c]	30
1/17/89	CT11C	150	0	15	35	252	0	50
1/18/89	CT12	231	3	10	15	284	5	60
1/18/89	CT12A	231	3	10	35	284	17	60
1/18/89	CT13	231	1	17	35	261	28	60
1/18/89	CT13A	231	2	15	35	262	7	45
1/19/89	CT14[d]	245	3	15	35	253	7	360
1/20/89	CT15A[e]	245	3	15	0	263	32	315
1/20/89	CT15B	245	3	15	0	338	126	60
1/21/89	CT16	150	2	15	35	247	15	60
1/21/89	CT17A	231	3	15	35	248	12	60
1/21/89	CT17B	231	3	15	35	256	53	60
1/24/89	CT18	231	3	10	35	261	22	67
1/24/89	CT19	231	3	10	35	319	38	59
1/25/89	CT20A	245	3	15	35	278	116	60
1/25/89	CT20B	150	2	15	0	310	72	75

[a] Full load rating is 231,000 lb steam/hr.

[b] SO_2 cocentration value is referenced to 12% CO_2 and is the continuous emission value.

[c] NA means not available.

[d] Sampling train for organics was located at the spray dryer inlet.

[e] Sampling train for organics was located at the fabric filter outlet.

TABLE 2. CHARACTERIZATION TEST RESULTS FOR FLUE GAS CLEANING SYSTEM OPERATED IN LOW SPRAY DRYER ABSORBER GAS OUTLET TEMPERATURE RANGE AND SEVERAL SO_2 CONCENTRATION RANGES AT FABRIC FILTER GAS OUTLET

FGC Test Series 1. Low Temperature (240-250 F)

SO2 Concentration at Fabric Filter Outlet	Low SO2 (0-20 ppm)				Medium SO2 (21-50 ppm)			High SO2 (> 50 ppm)				
CT No	11C	14	17A	18	11A	11B	6	3	8	9	10A	10B
Steam	161	244	207	143	161	149	163	151	216	231	153	192
Temp In	358	382	388	363	382	358	363	362	385	388	344	360
Temp Out	262	253	248	247	237	243	244	241	251	248	242	259
S Feed	11	6	4	5	11	11	NA	NA	6	8	0	0
Atm S	20	31	26	20	22	22	19	NA	29	31	22	22
SO2 In	138	192	173	136	168	158	171	170	170	182	160	136
SO2 Out	0	7	12	16	[illegible]	NA	44	108	207	186	236	201
% SO2 Removal	100	96	93	89	80	NA	74	38	NIL	NIL	NIL	NIL
HCl In	388	537	400	302	444	429	433	467	460	627	488	400
HCl Out	26	8	7	15	33	24	13	32	35	30	61	73
% HCl Removal	93	98	98	95	93	94	97	93	92	95	88	82

CT No. = Characterization Test Run Number

Steam = Steam Production Rate (klb/hr)

Temp In = Temperature of flue gas at spray dryer inlet (deg F)

Temp. Out = Temperature of flue gas at spray dryer outlet (deg F)

S Feed = Lime slurry feed rate (gpm)

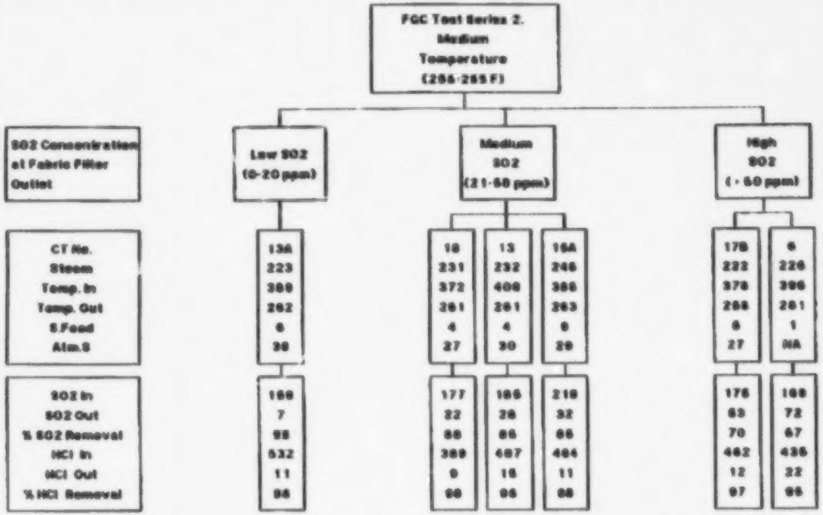
Atm S = Total flow (slurry plus dilution water) to atomizer (gpm)

SO2 & HCl In = Concentration at spray dryer inlet (ppm)

SO2 & HCl Out = Concentration at fabric filter outlet (ppm)

All concentrations are on a dry volume basis and referenced to 12% CO2.

TABLE 3. CHARACTERIZATION TEST RESULTS FOR FLUE GAS CLEANING SYSTEM OPERATED IN MEDIUM SPRAY DRYER ABSORBER GAS OUTLET TEMPERATURE RANGE AND SEVERAL SO2 CONCENTRATION RANGES AT FABRIC FILTER GAS OUTLET


FGC Test Series 2.
Medium
Temperature
(256-265 F)
SO2 Concentration at Fabric Filter Outlet
Low SO2 (0-20 ppm)
Medium SO2 (21-50 ppm)
High SO2 (> 50 ppm)
CT No.
Steam
Temp. In
Temp. Out
S.Feed
Atm.S
SO2 In
SO2 Out
% SO2 Removal
HCl In
HCl Out
% HCl Removal
13A 223 389 262 6 30
188 7 96 532 11 98
18 231 372 261 4 27
177 22 88 389 9 98
13 232 408 261 4 30
186 26 86 407 15 96
16A 246 386 263 6 29
218 32 85 404 11 98
17B 222 378 286 6 27
176 53 70 462 12 97
6 226 396 261 1 NA
169 72 57 436 22 95

CT No. = Characterization Test Run Number

Steam = Steam Production Rate (klb/hr)

Temp. In = Temperature of flue gas at spray dryer inlet (deg F)

Temp. Out = Temperature of flue gas at spray dryer outlet (deg F)

S.Feed = Lime slurry feed rate (gpm)

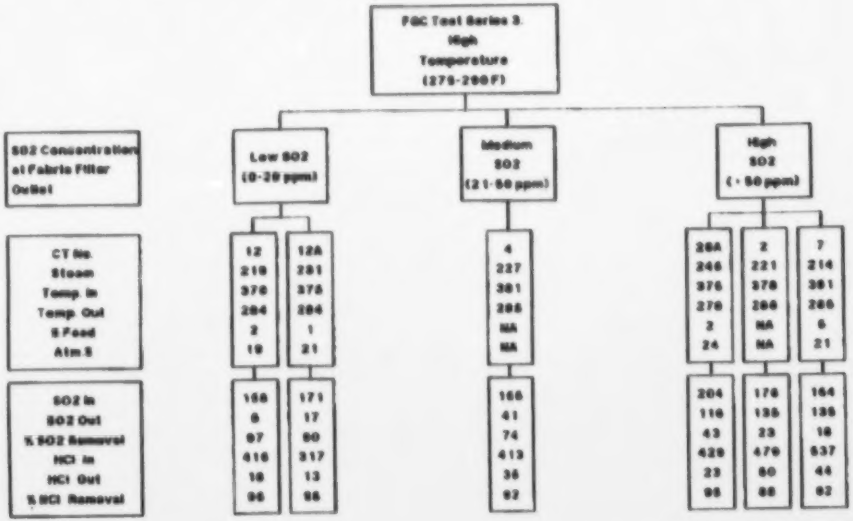
Atm.S = Total flow (slurry plus dilution water) to atomizer (gpm)

SO2 & HCl In = Concentration at spray dryer inlet (ppm)

SO2 & HCl Out = Concentration at fabric filter outlet (ppm)

All concentrations are on a dry volume basis and referenced to 12% CO2.

TABLE 4. CHARACTERIZATION TEST RESULTS FOR FLUE GAS CLEANING SYSTEM OPERATED IN HIGH SPRAY DRYER ABSORBER GAS OUTLET TEMPERATURE RANGE AND SEVERAL SO2 CONCENTRATION RANGES AT FABRIC FILTER GAS OUTLET

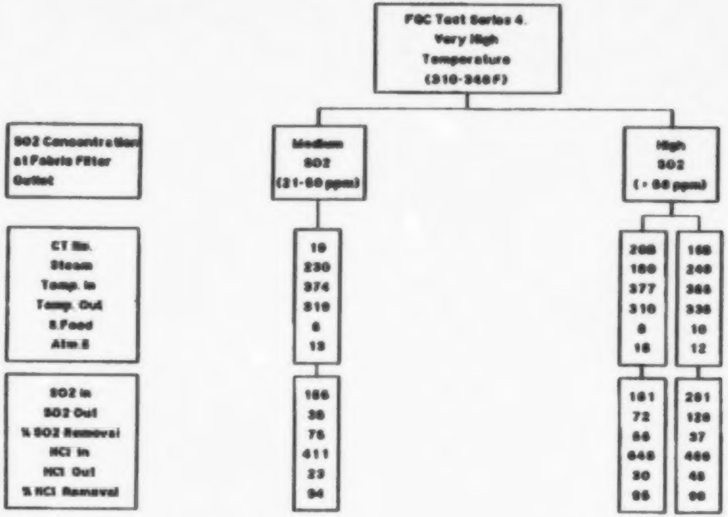


FGC Test Series 3
High
Temperature
(275-290 F)

SO2 Concentration at Fabric Filter Outlet	Low SO2 (0-20 ppm)		Medium SO2 (21-50 ppm)	High SO2 (> 50 ppm)		
CT No.	12	12A	4	26A	2	7
Steam	219	281	227	246	221	214
Temp. In	376	375	381	376	378	381
Temp. Out	284	284	285	279	286	285
S Feed	2	1	NA	2	NA	6
Atm S	19	21	NA	24	NA	21
SO2 In	158	171	156	204	176	164
SO2 Out	5	17	41	116	135	135
% SO2 Removal	97	90	74	43	23	18
HCl In	416	317	413	429	479	537
HCl Out	18	13	36	23	60	44
% HCl Removal	96	96	92	95	88	92

CT No. = Characterization Test Run Number
Steam = Steam Production Rate (klb/hr)
Temp. In = Temperature of flue gas at spray dryer inlet (deg F)
Temp. Out = Temperature of flue gas at spray dryer outlet (deg F)
S Feed = Lime slurry feed rate (gpm)
Atm S = Total flow (slurry plus dilution water) to atomizer (gpm)
SO2 & HCl In = Concentration at spray dryer inlet (ppm)
SO2 & HCl Out = Concentration at fabric filter outlet (ppm)
All concentrations are on a dry volume basis and referenced to 12% CO2.

TABLE 5. CHARACTERIZATION TEST RESULTS FOR FLUE GAS CLEANING SYSTEM OPERATED IN VERY HIGH SPRAY DRYER ABSORBER GAS OUTLET TEMPERATURE RANGE AND SEVERAL SO2 CONCENTRATION RANGES AT FABRIC FILTER GAS OUTLET


FGC Test Series 4.
Very High
Temperature
(310-340F)
SO2 Concentration at Fabric Filter Outlet
Medium SO2 (21-60 ppm)
High SO2 (> 60 ppm)

SO2 Concentration at Fabric Filter Outlet	Medium SO2 (21-60 ppm)	High SO2 (> 60 ppm)	
CT No.	19	208	158
Steam	230	180	240
Temp. In	374	377	366
Temp. Out	319	310	336
S.Feed	6	8	10
Atm.S	13	16	12
SO2 In	156	161	201
SO2 Out	38	72	126
% SO2 Removal	76	55	37
HCl In	411	648	466
HCl Out	23	30	46
% HCl Removal	94	95	90

CT No. = Characterization Test Run Number
Steam = Steam Production Rate (klb/hr)
Temp. In = Temperature of flue gas at spray dryer inlet (deg F)
Temp. Out = Temperature of flue gas at spray dryer outlet (deg F)

S.Feed = Lime slurry feed rate (gpm)
Atm.S = Total flow (slurry plus dilution water) to atomizer (gpm)
SO2 & HCl In = Concentration at spray dryer inlet (ppm)
SO2 & HCl Out = Concentration at fabric filter outlet (ppm)
All concentrations are on a dry volume basis and referenced to 12% CO2.

CHALLENGE OF THE 90'S -- MUNICIPAL WASTE COMBUSTION: THE PUBLIC-SOCIAL PERSPECTIVE

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ABSTRACT

Municipal waste combustion, as evidenced by numerous project cancellations and revisions to plans in recent years, is assuming a smaller role in waste management strategies nationwide than expected. The primary reasons are fear and lack of understanding about combustion on the part of the public and public officials. The challenge of the 90's is in the public arena where there must be an objective assessment of the role of municipal waste combustion in integrated solid waste management, that also includes reduction, recycling and composting. Polls show most people believe recycling will meet only about half the nation's waste disposal needs, but they oppose incineration and landfilling. Changing attitudes will require more familiarity with well operated facilities, appreciating the risks of all forms of waste management, and placing risk from combustion into proper perspective. Addressing public concerns through thoughtful information and meaningful public involvement is key.

For municipal waste combustion, the challenge of the 1990's is the public and social perspective, for it is clearly in the public and social arena that the momentum for waste-to-energy of the early 1980's was lost. Some 30 projects have been cancelled within the last 18 months. A rundown of one recent month's ironies sums up what we're dealing with:

> The Governor of New York, who is perhaps the Democratic party's most powerful orator, twice in a recent speech characterized burning, along with landfilling, as "necessary evils" (1). Ironic, since that very month, his environmental agency was promulgating strict landfill regulations and such tough emission regulations that the maximum calculated risk of exposure to dioxin emissions would be an infinitesimal 0.2 in a million (2).
>
> An environmental commissioner, whose state for several years had the highest standards in the East for waste burning, and has three plants that meet those standards, nevertheless, told a reporter, if he had to make the choice, he'd "rather live next to a landfill because it has no air emissions"(3). (Ironic, since USEPA's risk analysis, of which the commissioner was apparently unaware, tentatively calculates a maximum risk of 10,000 in a million from landfill emissions.) (4).
>
> "Don't Let City Garbage Go Up In Smoke" headlined an op ed piece in the New York Times (4) in which Dr. Barry Commoner, anti-incinerator crusader, railed against the proposed Brooklyn Navy Yard facility in New York, based largely on what he considers to be the "fact" that "incinerator ash is toxic and therefore not disposable in Fresh Kills landfill on Staten Island" (5). Again, ironic, since only a week before, a study by the firm that would build the plant showed that concentrations of lead, cadmium and mercury in combined incinerator ash are about the same as in urban soils, and that tests of actual leachate from such ash over 12 months show it essentially meets drinking water standards (6).
>
> Another New York Times article would give Dr. Commoner and his supporters cause to feel totally vindicated in their views (7). The article reported that a brand new plant in Warren County, New Jersey is going begging for waste because the garbage from other counties, tied up in contracts, is being hauled past the plant's door to out-of-state landfills. Recycling has reduced the amount of waste initially destined for the facility, and the plant's ash has to be disposed of as a hazardous waste because it fails a test procedure widely acknowledged to be inappropriate for ash. Ironic again.

> And from the midwest, Milwaukee's new mayor, who campaigned against incineration, exemplified the recent mind set in waste disposal planning by jettisoning any consideration of combustion in the city's Phase I plan in favor of what Milwaukee calls "3RC" -- reduction, reuse, recycle and compost. Ironic because the mayor's own citizen advisors had labored for months fashioning the scope of an integrated waste management study that included combustion as an option.

Indeed, the conference we are now attending is the first in recent memory dedicated to the "C" word -- Combustion, with a capital "C" -- as opposed to the popular approach we've come to refer to as "integrated waste management". The fact is that at the current rate, there will be less and less integration of combustion into "integrated" waste management strategies, and more and more reliance on totally non-combustion approaches -- the 3R's and composting -- unless totally non-combustion approaches prove unrealistic for some areas of the country.

Clearly, the potential for non-combustion 3RC techniques is greater than most people ever believed possible. Newark, New Jersey, for instance, a city with an unfair measure of urban ills -- littered streets, homelessness, drugs, struggling schools, unemployment -- is nevertheless reported to be recycling better than 30 percent per person (8). Rhode Island reports high enthusiasm for recycling crossing all socio-economic strata (9). And perusal of recent issues of *Waste Age* or of the booths at an NSWMA trade show suggests there is even more entrepreneurial energy behind 3RC than there is behind waste-to-energy.

But will the 3RC revolution do more than offset the continuing growth of waste? Perhaps, if markets develop fast enough to absorb the growing tide of post-consumer scrap materials, and education truly changes consumer and producer behavior -- and sustains it. But what is more likely is that serious reduction and recycling will be needed just to stay even. In a recent study for USEPA by Franklin Associates (10), the conclusion was that even if anticipated recycling *and* energy recovery are accounted for, net discards nationwide will decline only six percent by the end of the century, from 2.98 lbs per person in 1986 to 2.8 lbs in the year 2000.

I'm the first to admit to a dilemma. Those of us who actively supported recycling and waste reduction as early as the 70's because it was the environmentally correct thing to do, find it bittersweet now to see so many people embracing these measures largely out of political expediency, fear and ignorance of the realities of municipal waste combustion.

On the one hand, three national polls taken in 1988 for NSWMA revealed that Americans, overall, place garbage disposal second only to improving education as the most serious problem facing local officials; in states north and east of Illinois, it's the number one problem (11). On the other hand, while most people believe that no more than 50 percent of our waste can be recycled, 47 percent of the respondents nationwide oppose building waste-to-energy plants. Even in the Northeast, where garbage disposal is regarded as far and away the top local issue, 49 percent share this view. Only landfills fare worse; 65 percent of the people polled are against them.

Clearly, without major changes in attitudes about waste-to-energy, we are at an impasse. But are such changes of attitude feasible when public hysteria routinely overrides judgment and science, and when exhaustively prepared analyses of health effects, concluding that the maximum risk of cancer from waste-to-energy facilities is negligible, are misunderstood by officials and the public or, worse, dismissed out of hand by knee jerk opponents?

In today's environment, local officials, even those who have read and mastered the complex information, must display extraordinary political and personal courage to stand up before truly threatening crowds on the strength of their convictions alone. They also need political and intellectual stamina. No project will proceed without a determined long-term commitment by the project sponsor. This is not the arena for pulse-taking politicians, hesitant entrepreneurs, or under-funded projects.

Perhaps the national phobia over combustion will subside when more people get to "kick the tires" of longer-operating plants, get to talk to neighboring residents and businesses, walk on an ashfill, and familiarize themselves enough with the material to believe the official test reports on the operation of these facilities. Perhaps combustion will be viewed more objectively when the risks of all forms of waste disposal are put into perspective.

The City of Milwaukee made a good beginning in evaluating the risks of an integrated solid waste plan (12). The authors of the plan point out that the difference between a 3RC program and an incinerator or landfill is that the former tends to distribute the risks broadly (to workers, the general public, and distant populations) rather than to the locality of the refuse disposal facility. Diffusing risk, even if in total the risk is greater, has undeniable political appeal. Examining the risks of the entire waste management system is an obligation of all solid waste planners, especially USEPA. Jurisdictions embarking on comprehensive solid waste planning should build on the Milwaukee study.

Another approach is to put the environmental impact from a proposed waste-to-energy facility itself into perspective. Understanding will be advanced, for instance, by pointing out that the hourly particulate emissions of 1.25 lb/hr from a 1,000 ton per day facility with a scrubber and baghouse are equal to the hourly particulates from 40 household-size non-catalytic wood stoves meeting EPA's Phase I limit (0.03 lbs/hr), and by pointing out that household chimneys, unlike taller stacks, disperse emissions poorly. Or, that the maximum calculated risk from a resource recovery plant of less than one-in-a-million is at least 460 times less than the lifetime risk from as routine an activity as daily showering with chlorinated water, as calculated by EPA (13). Or, as noted earlier, that the one-in-a-million risk from resource recovery is 10,000 times less than the risk of the maximum exposure to landfill emissions.

But someone has to make sure that this information is communicated, especially to the core group of local representatives who are willing to make the effort to understand the basis of such estimates. Because so few local officials read anything more than their daily newspapers, sensational claims in the press are frequently more persuasive than consultant reports. That's got to change.

A major step toward increasing confidence in the safeguards built into modern combustion facilities would be greater public awareness of the rigors of the regulatory process. People are so disaffected by the enforcement performance of environmental agencies on existing problems that they have little appreciation for the intense scrutiny that permit applicants must undergo and the strict demands they must meet. This knowledge would help decision makers and local officials feel they are embarking on a safe and economically prudent course of action.

To support that conviction, they will often turn to public interest advisors. Several projects have survived challenges because a citizens group was involved in each step of the process and concurred with decisions as they were made. Ideally, citizen advisors would be involved in determining the need for a facility, and in considering the technology, sizing, site selection, environmental and health impact, and special operating conditions of a facility. If any of these steps are not sufficiently considered, and the decisions are not thoroughly justified, charges will continue to haunt the project until construction gets underway -- or until the project is terminated. Of course, in order to be constructive participants, citizens cannot operate on the basis of preconceptions. They must be informed. One effective means is to make professional consultants fully available to the advisory group.

Citizen involvement is producing better projects in many places. Today's definition of Best Available Control Technology for municipal waste combustion was first articulated by a citizen advisory committee to the Brooklyn Navy Yard facility in New York. Citizen pressure for recycling is resulting in downsizing projects and establishing incentives to remove troublesome materials, like batteries, prior to collection. And use of citizen-approved siting criteria has resulted in acceptance of some facilities when such a public siting process has been applied.

It may, in fact, be only through thoughtful citizen examination of issues that public officials will avoid falling prey to some popular beliefs about waste that have little environmental validity or disposal benefit. Take, for example, the push toward programs for separate collection of household hazardous waste (HHW) when these materials could otherwise go to a municipal waste combustor. (Collecting HHW is more valid when a landfill is the only disposal option.) If provided with objective information, citizens would very likely come to appreciate that the risks arising from inexperienced people storing, transporting and handling mixtures of hazardous household products to a drop-off site are more real than exposure to low emissions from combusting these organics in well-operated waste-to-energy plants. They might also decide that, at an estimated cost of $18,000 per ton for collection and disposal of household hazardous waste (14), the economics of such programs don't stand up.

Perhaps the foremost challenge of the 90's is to develop print and audio-visual materials that speak directly to the issues on people's minds, and to encourage -- train, if necessary -- effective spokespersons who can address the whole spectrum of public concerns over waste combustion in language that commands as much public attention as the rhetoric of the charismatic opponents of incineration. We need articulate spokespersons, who know garbage, who know the technical issues, and who know how to talk to people.

Entering the 1990's, combustion is viewed by the general public as, at best, a "necessary evil." By beleaguered officials, it is a difficult path to be avoided if possible. The pendulum has swung far over, and clearly, a certain amount of natural swinging back will have to take place. But, municipal waste combustion will fulfill its legitimate role in solid waste management in the 1990's _if_ it is seen as the companion to aggressive programs to reduce and recycle as much waste as possible, _if_ operating facilities continue to be good neighbors and meet permit conditions (as they do in most cases), _if_ the public has confidence in the regulators, and _if_ project sponsors invest the resources in building informed public consensus for inclusion of municipal waste combustion in truly integrated solid waste management programs. Such integrated programs will consist of 3R _double_ C -- Recycling, Reuse, Reduction, Composting _and Combustion_.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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TECHNOLOGICAL OUTLOOK FOR THE '90'S:
THE SUPER TECH CHALLENGE

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ABSTRACT

The advance of waste-to-energy technology in the 1980's has been slowed by a lack of comprehensive technical data for issues of concern. The consequences have been uncertainty, confusion, and fear among regulatory bodies, project developers, and local populations, and subsequent project delays, cancellations, and cost increases. The historical approach to dealing with technical uncertainty has been characterized by a Super Fund/Super Regulation mentality which provides reactive decisions based on piecemeal research as well as highly conservative regulatory requirements to compensate for technical uncertainty. The issues of the 1990's must be confronted with a proactive Super Tech mentality, characterized by a major national commitment to scientific research and development. The Super Tech approach is essential to find solutions for the solid waste disposal crisis and address the overall objective of protecting public health and environmental safety.

INTRODUCTION

The challenge of the 1990's is to begin to base decisions regarding solid waste management on a solid foundation of scientific fact, rather than on the uncertainties, confusion, and fear which have characterized the 1980's. Decision-makers have been compelled to base far-reaching decisions on piecemeal research and a general vacuum of knowledge which has been supplanted by emotion and secret agendas. The advance of solid waste management technology in the 1980's has been ham-strung, if not totally crippled, by microscopic glances at fragments of issues, driven by competing special interests. What is needed in the 1990's is a hard, scientific gaze at the macroscopic issues of solid waste management and environmental protection. Without this scientific perspective, we can expect unnecessarily high disposal costs, continued dependence on municipal landfills, and failure to address the larger objective of protecting public health and environmental safety.

At least three key issues evolved in the waste-to-energy (WTE) industry in the 1980's that were characterized by uncertainties and punctuated by fear in the absence of reliable, comprehensive information: dioxins in stack emissions, heavy metals in ash residue, and ammonia injection as a potential mandated requirement for control of nitrous oxide (NO_x) emissions. Historically, we have met environmental issues with a Super Fund mentality, characterized by spending big money to remedy past mistakes. The Super Fund mentality begat Super Regulation, characterized by demands for highly conservative and often prohibitively expensive technical requirements to compensate for technical uncertainties. When the questions and fears regarding impacts of dioxin and ash were not met with expeditious answers, WTE project opponents demanded Super Fund/Super Regulation solutions.

We must evolve beyond the Super Fund/Super Regulation mentality and develop a proactive problem-solving approach driven by a Super Tech mentality. The Super Tech approach demands a major national commitment to scientific research and development to attack future technical and environmental concerns as they emerge, reducing and eliminating prolonged uncertainty and fear and the reactive decisions they generate. Our mentality of big spending for big cleanup must evolve to a scientific mentality of funding research to solve big challenges.

THE DIOXIN DEBACLE

Dioxin is the commonly used term representing an entire family of organic compounds, some of which are known to be extremely toxic to some animals. Human exposure to dioxin has occurred from accidents at chemical manufacturing plants, such as the explosion at the Seveso, Italy plant in 1976; occupational exposure to herbicides over a long period of time, including military personnel exposed to Agent Orange during the Vietnam War; and contamination of the general environment which exposed local residents to low doses of dioxin over a long period of time, such as the Times Beach, Missouri soil contamination. Research regarding these types of human exposures has not demonstrated any statistical correlation between dioxin exposure and long-term human health effects, although research in this area continues.

During the late 1970's, dioxins were identified in the stack emissions of several European, Japanese, and U.S. WTE facilities, and quickly became a subject of concern. Health risk assessments were conducted to assess the potential health impacts from dioxin exposure. The lack of data regarding human health impacts from dioxins compelled scientists to incorporate many worst-case assumptions into the health risk assessments, which yielded rates of risk which seemed unacceptable. The lack of knowledge regarding the quantities of dioxins which would be emitted and their potential impact on human health precipitated public fears regarding siting of WTE facilities in their communities. It is likely that the uncertainty, fear, and confusion regarding dioxin emissions caused cancellation of some WTE projects, and significantly increased costs and delays of others.

During this prolonged period of uncertainty, a body of data gradually emerged which showed that technological innovations such as dry scrubbers and combustion control techniques reduced dioxin emissions significantly. Health risk assessments, which continue to incorporate very conservative assumptions in the absence of knowledge in certain areas, now indicate that the risks to the population from WTE dioxin emissions are minimal, considerably less than risks posed by many everyday activities.

It is comforting that the data now indicates that WTE dioxin emissions should pose minimal risks to human health. However, the prolonged period of fear and confusion cost us in terms of continued reliance on landfilling. It is questionable whether reliance on landfilling fulfills our overall objective of protecting public health and environmental safety since many existing landfills have been shown to be polluting groundwater with leachate seepage; may be breeding grounds for rats, flies, and other vermin; and may pose health and safety risks due to emissions of gases such as methane or trace compounds such as vinyl chlorides.

The Super Tech approach would have expedited research regarding the potential risks of dioxin, seeking to identify the nature and extent of the problem and propose courses of action. Super Tech would seek to dissipate the mists of uncertainty as quickly as possible to allow regulatory bodies, project developers, and local populations to base decisions on solid scientific truths.

THE HEAVY METAL HAMMER

One of the products of the Resource Conservation and Recovery Act (RCRA) of 1976 was the creation of certain tests intended to evaluate the characteristics of suspected hazardous wastes. The Extraction Procedure (EP) Toxicity test was intended to simulate the leaching a waste would undergo if disposed of in a municipal solid waste landfill. During the mid-1980's, questions were raised regarding whether WTE facility ash was subject to EP Toxicity testing prior to disposal. Despite certain legislative exclusions, EPA at one time stated that if WTE ash failed the EP Toxicity test, then the ash should be managed as a hazardous waste. Many state regulatory bodies followed suit.

Some WTE ash samples failed the EP Toxicity test due to high levels of lead and cadmium. Flyash, if segregated, consistently failed the test for these metals. Project opponents cited these results as evidence that WTE ash should be managed as a hazardous waste or it could pose risks to human health. Confusion and controversy erupted regarding a multitude of ash testing issues. How should the ash be sampled? How many samples should be tested? How do you resolve uncertainty when two laboratories fail to produce the same test results for the same ash sample and a single lab may produce different test results for a single ash sample? Should the flyash and bottom ash streams be segregated or is mixing a good thing?

Some research indicated that the EP Toxicity test does not simulate the leaching conditions in ash monofills, and consequently would be an inappropriate test to evaluate the characteristics of ash. However, questions regarding what would constitute a useful testing protocol remain unanswered.

Is reuse of ash technically feasible and environmentally sound? Is treatment necessary and would it offer significant environmental benefits?

Most of these issues have yet to be resolved. The body of data regarding the characteristics of ash is still inadequate to definitively answer these questions. The unfortunate consequences of these uncertainties are that sorely needed plants have likely been cancelled, new projects have met strong opposition, and existing plants have also been perceived as a health threat by the public. Some plants currently face expensive testing requirements to conduct the questionable EP Toxicity test required by some states, and at least one plant has been forced to shut down due to the high costs of hazardous waste disposal as a consequence of failing the EP Toxicity test.

Once again, our efforts to confront the solid waste disposal crisis have been slowed by technical uncertainty, confusion, and fear. While we wait for the answers, we continue to consume our diminishing land resources, potentially jeopardizing our overall environmental safety from landfilling which poses its own leachate risks.

The Super Fund/Super Regulation response to managing the uncertainties surrounding ash has been to consider potentially expensive lined monofills, hazardous waste disposal, and other conservative measures to compensate for the lack of technical knowledge. There are a number of valuable research programs underway, such as those being conducted by the New York State Energy Research and Development Authority (NYSERDA) and the University of Connecticut which should increase our knowledge of the characteristics of ash. Curiously, other attempts to gather information have been stifled. In Connecticut, for example, the owner of a major WTE facility and virgin landfill site offered to completely fund a research project wherein scientific data could be collected on monofilled ash leachate for various soil and liner designs. The Connecticut Department of Environmental Protection (DEP) denied the permit, demanding that only double plastic liners were to be used. Efforts such as these which are rejected prolong the vacuum of knowledge on which policy and regulation will be based. Super Tech would have embraced the ash issue and marshalled the resources necessary to attack the uncertainty.

THE NO_X BOX

One of the most recent trends is toward mandated installation of ammonia injection on new WTE plants for nitrous oxide (NO_X) emissions control as best available control technology (BACT). Ammonia injection has been shown to reduce levels of NO_X in stack emissions. However, technical issues such as potential boiler tube corrosion and air pollution control equipment corrosion have not been resolved. Since NO_X has not been shown to be a significant health concern, the Super Fund/Super Regulation approach toward mandating ammonia injection could be premature. Such a mandate could stifle other innovative solutions to controlling NO_X emissions and could precipitate potentially expensive modifications to prevent accelerated equipment deterioration. If control of NO_X emissions is to become a challenge of the '90's, the Super Tech approach will step forward with proper funding for research into ammonia injection before a final decision is made.

OUTLOOK

The '90's can become the decade when the real engineering of solutions begins, based on scientific fact. The '80's provided for individual technology demonstrations drawn from trial and error experiences in the real world laboratory. Now it is time to combine the individual building blocks into blended technological solutions, where alternatives complement, not compete.

While WTE issues have been discussed in this paper, WTE technology is not alone. The Super Tech mentality is essential to enable a transition from management of individual infrastructure residuals to combined residual streams. We can expect a shift to consolidation of our management of residuals such as sludge and ash, as well as municipal solid waste. Comprehensive programs could include combination of curbside yard waste collection with disposal of wastewater and water treatment sludges to comprise a co-composting operation, or combination with refuse-derived fuel for co-incineration.

Continued emphasis can be expected on recovery of resources from total municipal residuals either through separation or conversion. Energy recovery will continue as it recovers from the Super Fund/Super Regulation uncertainty and hesitation, but additional markets for other products will develop and stabilize. Secondary fiber utilization in the pulp and paper industry is a prime example. Management processes and technological tools will be needed to recover waste stream components as manufacturing feedstocks.

New problems and challenges will arise in the 1990's, but they will only be solved when we commit strongly to a Super Tech program of comprehensive scientific research and development. In an article written for the Washington Post, entitled "Holes in the Greenhouse-Effect Theory," Mr. Patrick Michaels, in a call for balanced scientific debate as opposed to emotionalism, concludes:

> "Our policies should be no more drastic than the scientific conclusions they are based upon . . . . Otherwise, we may very well be about to make a policy blunder of literally global proportions. But what I fear is that all of the recent findings will be ignored. Policy - especially when prompted by very powerful special interests - has an inertia that is very difficult to overcome once the ball starts rolling, which it most certainly has. That makes all the hard work, all the science, inconsequential and irrelevant to those who say: Damn the data, full speed ahead!"

Technological outlook for the '90's? The challenge is to implement Super Tech. We must seek sources of funding at federal, state, and local levels; combine and focus research efforts by public and private entities; and distribute information to regulatory bodies, project developers, affected communities, etc. Super Tech demands that we fund research and development and develop a focused, aggressive approach to attacking new issues to dispel the mists of uncertainty and achieve our common goals.

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ECONOMIC TRADEOFFS IN MUNICIPAL WASTE COMBUSTION

Good morning, my name is Warren Gregory and I'm with Merrill Lynch. I have been asked to discuss economic "trade-offs" in municipal waste combustion. My perspective is that of the banker who is seeking the lowest financing costs for the project while trying to minimize the risks of the political subdivisions who will be using the project for waste disposal. Although some facilities have achieved lower costs by having the municipalities bear most of the projects economic risks, this "trade-off" is not a good one in most instances since the reduction in interest rate does not substantially lower the cost per ton of the project on a present value basis.

Another way to express this might be to say that if you have a hundred basis points of savings on a one hundred million dollar project cost that processes about 300,000 TPY you would save $2.64 per ton in interest for the lower interest rate. In comparison to the same project that was negotiated for a 10% increase in net electric output per ton which at .07 cents /kwh and a 90/10 sharing would give the community about $3.15 per ton more in energy revenue credit. This may be a simplistic example but one that has been lost on many projects. Too often the community perceives its efforts to be one of risk control rather than solid waste disposal. Whatever its structure, if the project does not work, the community has lost a great deal of effort and time. Even if it is insulated from the actual debt service repayment, it has lost the confidence of the electorate and the site itself, since the trustee will negotiate for the bond holders not the city. The early 70's saw a host of projects that promised more than they could deliver and delivered more than anybody wanted in problems and failures.

We try to convince our public clients to start with the premise that although the solid waste industry may be operated by private companies in their area (as it is in most parts of the United States), the responsibility for the public health and safety will always rest with the elected official. When the garbage piles up on the streets, the costs of trying to create a system that shelters the public from all the risk is impractical, and in the final analysis, more costly than the balanced approach.

A good starting point is to try to divide the risks of a project by defining who can best be responsible for them. Technical risks, as well as the long term operating risks, should be born by the builder. The risk that future laws will effect the project should be born by the participating communities since they make those laws and can most effectively influence their outcome. These are practical and seemingly commonsense divisions that get lost in the process of project development. Those of us who work in this area must be willing to identify these alternatives and ask our clients to carefully consider them. Sometimes this must be balanced against the competitive forces of the market place and the political realities of the particular jurisdiction. Most of the men and women who specialize in this area of finance have done a credible job keeping their clients well informed. However, some jurisdictions do not bring an investment banker into the project team until after the risk structure has been defined in the RFP process. This, in turn, makes it difficult for the banker to have any input since the RFP document has already been released and changes are usually complicated and costly. I strongly recommend including the banker before the RFP is released and making the banker responsible for the credit structure of the project. This means that the decisions are made up front and not at the end of the financing process where the delays are much more costly and difficult.

Another major area of "trade-off" is ash disposal. Most communities do not have facilities sited for this product – and most would not like to address this siting issue. Understandably, for it takes a strong and far sighted elected official to propose a waste-to-energy facility given the placebo of other alternatives. A "trade-off" that has worked in many projects has been to make the plant operator responsible for the quality and quantity of the ash and the ultimate disposal in an environmentally sound manner while the community is held responsible for the costs on a "pass-through" basis with the option of finding their own site if the alternative found by the vendor is not the most economical or practical.

Hazardous waste also presents another economic "trade-off:. It often seems more difficult than it really is because of the heightened sensitivity the subject has in the press. Most municipal waste has little hazardous materials -- a minute percentage of the total volume. However, the operator must be watchful and vigilant so that these occasional waste items that are illegally dumped are not incinerated but rather removed from the tipping floor and sent to a hazardous waste site. This discussion usually starts with the Catch 22: of: "There's no hazardous waste in the municipal waste we collect so if its in your plant it got there by another method." This is countered by the indignant operator who tells you that the combustion process does not create hazardous materials that such materials must have never been in the waste when it was delivered. Both of course are right. So let's go back to the first principal of risk separation and make the person most able to bear the risk responsible. If the operator can spot the waste when its dumped and stop the truck, it goes back on the truck and out the door with that truck -- his risk. Alternatively, if the waste is located after the truck has left but it can be identified by its container or markings, then the manufacturer gets the call and the bill. If, however, it can't be identified and the truck has gone, then the plant operator must bear this cost.

Another important "trade-off" consideration is that of systems financings. We began this discussion by reviewing the "trade-offs" that are project specific. It should be emphasized that although these are important considerations, the real purpose of a waste-to-energy combustion facility is volume reduction to preserve landfill and the potential environmental impacts associated with the disposal of raw garbage. The community recycling system must also be a key part of this financing since it will shape the quality and the content of the waste streaming for years to come.

Recycling systems also represent the greatest real test of source reduction savings since waste that is not delivered to the landfill or the resource recovery facility must be accounted for by the system to reflect the true overall cost of wasted disposal.

I believe that the future financings will reflect a pricing system that included all these factors. We cannot have a waste management system financing that does not encompass all the costs required to completely dispose of waste in an environmentally sound manner or that does not return materials to the stream of commerce.

Thank you.

PILOT-SCALE ASSESSMENT OF NATURAL GAS REBURNING TECHNOLOGY FOR NO_x REDUCTION FROM MSW COMBUSTION SYSTEMS

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ABSTRACT

A program is under way to evaluate the use of natural gas as an NO_x control technique for municipal solid waste (MSW) combustion systems. The program is funded jointly by the Institute of Gas Technology (IGT) Sustaining Membership Program (SMP), the Gas Research Institute (GRI), and Riley Stoker Corporation (Riley) and is a cooperative developmental effort between IGT, Riley, and Takuma Company, Ltd. (Takuma). The program entails assessment of applicable NO_x control technologies and acquisition of baseline data from a modern existing MSW mass burn facility, followed by pilot-scale development of the selected NO_x control techniques and preparation of design specifications for field verification of this technology. This paper concentrates on the results of the pilot-scale developmental efforts.

The technology selected for evaluation is natural gas reburning (NGR) — which primarily involves injection of natural gas into the products of MSW combustion to provide reducing conditions for the destruction of NO_x, as well as of NO_x precursors. Overfire air (OFA) is then injected downstream to burn out the remaining combustibles. Combined with flue gas recirculation (FGR) and improved mixing in the OFA region, NGR is expected to reduce NO_x to below 50 ppm (at 12% O_2) while improving the OFA combustion stability and uniformity and decreasing the overall excess air requirement. This should provide enhanced combustible burnout and improved system thermal efficiency.

Following extensive testing of a commercially operating Riley-Takuma MSW mass burn system for baseline data acquisition, NGR was evaluated in IGT's

pilot-scale furnace at a rate of 1.7 X 10^6 Btu/h using simulated combustion products that would result from firing of MSW. Effects of residence time, stoichiometry, combustion products temperature, MSW nitrogen content, amount of natural gas, and level of FGR on gaseous emissions were investigated.

Further development of the NGR technology is being carried out at Riley's 6 ton/day (3 X 10^6 Btu/h) MSW combustion system that includes scaled Takuma grates and Riley's waterwall furnace. This flexible system is being used to develop the best combination of natural gas, recirculated flue gases, and OFA injection strategies to achieve the desired emissions reduction. This paper discusses the potential environmental as well as operational benefits of NGR technology in MSW combustion systems based on the results of the pilot-scale developmental efforts at both IGT and Riley.

INTRODUCTION

On any given day in the United States, a total of about 450,000 tons of municipal solid waste (MSW) is generated — of which a majority (90%) is landfilled (1). Space is running out, however, and more and more municipalities are looking for alternative disposal techniques. One such alternative is incineration, which has been gaining popularity in recent years, especially since it allows a considerable amount of heat recovery while being environmentally sound. It is estimated (1) that by the year 1992, almost one-fourth of the MSW may be processed. Technology for the mass burning of MSW has been developed and utilized in Europe and Japan for decades. This technology has been adapted to U.S. needs and has become the basis for the majority of the current U.S. MSW Resource Recovery Plants. Significant advances, considering the relatively poor quality of the MSW fuel, have been made in combustion, heat recovery, and environmental control equipment. Existing plants, however, still experience some problems including air emissions. More and more stringent regulations are being formulated and imposed to limit these emissions.

The air emissions generally encountered from waste-to-energy facilities are similar to those from most combustion operations: oxides of nitrogen (NO_x), oxides of sulfur (SO_x), carbon monoxide (CO), total hydrocarbons (THC), and particulates. Because of the relatively low sulfur content of MSW, the SO_x emissions are proportionately low. The presence of large quantities of chlorine in the range of 0.6% raises concerns about hydrogen chloride (HCl) emissions as well as about the potential for production of trace polychlorinated hydrocarbons such as dibenzodioxins (PCDD) and dibenzofurans (PCDF). In addition, certain metals in MSW can be vaporized during combustion. Most of them condense and solidify later, as combustion products are cooled. Most modern facilities are equipped with lime treatment for HCl and SO_x removal and with either baghouses or electrostatic precipitators for capturing particulates from the flue gases. Further, high excess air levels of up to 100% are employed to ensure low CO and THC emissions. The literature (2) implies that low emissions of CO should also result in low production of polychlorinated hydrocarbons.

The primary emissions of concern in this program are NO_x. With typical levels of 150 ppm,[1] the NO_x produced from MSW facilities could approach 150,000 tons annually in the near future. Significant work has been undertaken, particularly overseas, on NO_x abatement in MSW facilities. Work done by Nippon Kokan K.K. (3) has indicated the potential of NO_x as low as 80 ppm by combustion modifications. Other techniques such as water injection, flue gas recirculation (FGR), thermal de-NO_x, and selective catalytic reduction (SCR) have also been evaluated with some degree of success (see Table 1).

This paper reports the results of a program that has been initiated to evaluate the use of natural gas as an NO_x control technique for MSW facilities. The technique selected is natural gas reburning (NGR), which is schematically illustrated in Figure 1 using a sloped stoker mass burn combustor as an example. It essentially entails injection of natural gas into the products of MSW combustion to create reducing conditions. These conditions promote destruction of NO_x as well as decomposition of other fixed nitrogen species such as NH_3 and HCN (that have already formed in the primary combustion zone) to harmless molecular nitrogen. After sufficient residence time is allowed at the reducing conditions, overfire air (OFA) is then injected downstream to burn out the remaining combustibles. Combined with FGR and improved mixing both in the NGR as well as in the OFA region, NGR technology is expected to significantly reduce NO_x with the added benefits of enhanced combustible burnout. Significant reductions in overall excess air levels and improved system thermal efficiency are also expected. Table 1 summarizes the potential for the NGR technique as compared to other proposed technologies. The NGR technique appears quite attractive.

The multi-phase technology development effort described in this paper entails acquisition of baseline data from an existing mass burn facility, followed by assessment of NGR on a pilot-scale furnace by simulating products from MSW combustion, evaluation of NGR strategies on a pilot-scale mass burn facility using processed MSW, aerodynamic evaluation on a one-twelfth-scale cold flow model for scale-up data, preparation of design specifications for field testing, and finally, field testing of the advanced NGR technology.

In this paper, we will summarize the baseline field testing, discuss in detail the results of pilot-scale testing using simulated MSW combustion products, and review the status and the available results of the pilot-scale tests using processed MSW.

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[1] Concentrations of NO_x, CO and THC in this paper refer to a 12% O_2 basis,unless specified otherwise. To convert to the 3% O_2 basis, if necessary, multiply by 2.0.

Table 1. COMPARISON OF AVAILABLE NO_x EMISSION REDUCTION TECHNIQUES

	Water Injection	Combustion Modification	FGR	Urea (Ammonia) Injection	Proposed NGR	SCR
Expected NO_x, ppm	80	80	80	60	50	15
Other Effects	Lowest Cost, Higher CO Emissions, Thermal Efficiency Loss	Higher Initial Cost, Higher CO Emissions	Higher Initial Cost, Higher CO Emissions, Thermal Efficiency Loss	Higher Initial and Operating Costs, Urea (Ammonia) Emissions	Higher Initial and Fuel Costs, Lower CO Emissions, Thermal Efficiency Gain	Very High Initial and Operating Costs

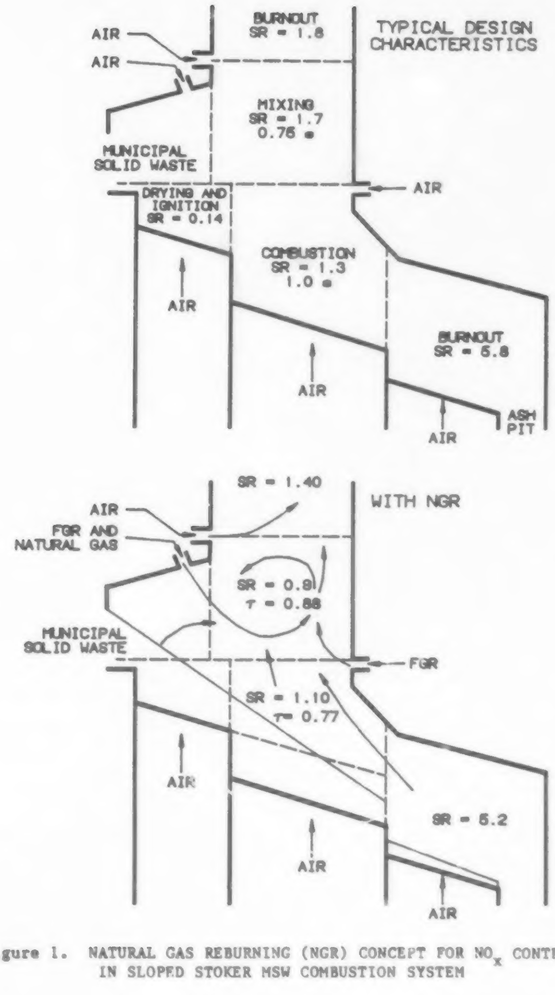
AIR
AIR
BURNOUT
SR = 1.8
TYPICAL DESIGN CHARACTERISTICS
MIXING
SR = 1.7
0.76 s
MUNICIPAL SOLID WASTE
DRYING AND IGNITION
SR = 0.14
AIR
COMBUSTION
SR = 1.3
1.0 s
AIR
AIR
BURNOUT
SR = 5.8
AIR
ASH PIT
AIR
SR = 1.40
WITH NGR
AIR
FGR AND NATURAL GAS
SR = 0.9
τ = 0.88
MUNICIPAL SOLID WASTE
FGR
SR = 1.10
τ = 0.77
AIR
SR = 5.2
AIR
AIR

Figure 1. NATURAL GAS REBURNING (NGR) CONCEPT FOR NO_x CONTROL IN SLOPED STOKER MSW COMBUSTION SYSTEM

BASELINE FIELD TESTING

FACILITY DESCRIPTION, TEST ARRANGEMENTS, AND TESTING

The baseline data were acquired on one of two units at the Olmsted County Waste-to-Energy Facility located in Rochester, Minnesota. The design of the combustor is an integration of Takuma Company, Ltd. (Takuma) MSW stoker and combustion control technology with Riley Stoker Corporation (Riley) waterwall furnace technology. The design details have been described earlier (4). Each unit was designed to burn MSW at the rate of 100 ton/day producing about 24,000 lb/h of 615 psig superheated steam. The steam is used to heat a nearby Federal prison and/or to drive a turbine.

The unit was tested while varying load, total stoichiometric ratio (TSR), allocation of undergrate air (UGA) flow, and OFA location. Two general types of testing were conducted: 1) in-furnace measurements by the Institute of Gas Technology (IGT) and 2) overall system performance data acquisition by Riley.

The in-furnace data were collected through available viewports of the furnace, either near the top of the refractory lining or at the "furnace exit" which is in the bend between the first and second radiant sections of the furnace. These locations are shown on Figure 2. Also, during a prolonged test at tuned operating conditions, gas composition data were acquired above the three stoker grates, but below the OFA zone.

TEST RESULTS

Details of the baseline field testing and results have been published earlier (5). Some of the more significant results, primarily on NO_x emissions, are summarized here. The NO_x produced from the facility ranged from about 125 to 175 ppm when the facility was operating normally — 100% load, over 60% excess air, and at least two OFA sets operating. NO_x was generally lower at lower excess air and with less than two OFA sets.

Figure 3 compares the NO_x concentrations measured in traverses (in duplicate) at the furnace exit with the corresponding NO_x concentration at the precipitator inlet. It is apparent that any NO_x produced is stable in the system. Table 2 summarizes the impact of OFA mixing on NO_x, CO, and THC emissions. With normal operation, the NO_x was in the range of 125 to 175 ppm and the CO and THC concentrations were already reduced to <30 and <15 ppm, respectively, at the furnace exit. At the same overall excess air level, but with all the air supplied under the grates, the NO_x emissions dropped by 30% to 88 ppm, but both the CO and THC emissions increased. With still no OFA, and the overall excess air reduced to 23%, the NO_x emissions dropped to 65 ppm — a 54% reduction — over the normal value. Both the CO and the THC concentrations, however, shot up to 50 times the normal values. The amount and distribution of the combustion air, therefore, plays a significant role in determining the emission levels from these facilities.

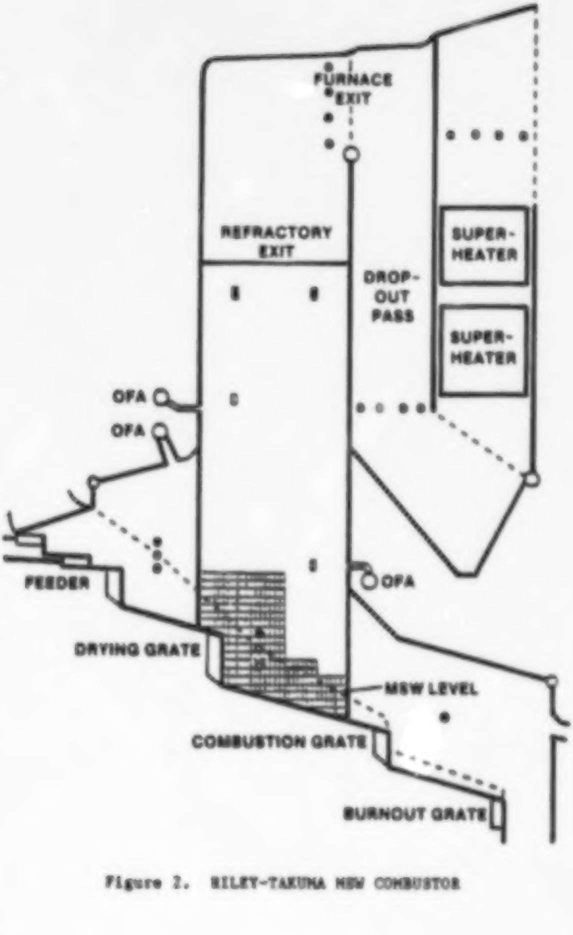
FURNACE EXIT
REFRACTORY EXIT
DROP-OUT PASS
SUPER-HEATER
SUPER-HEATER
OFA
OFA
OFA
FEEDER
DRYING GRATE
MSW LEVEL
COMBUSTION GRATE
BURNOUT GRATE

Figure 2. RILEY-TAKUMA MSW COMBUSTOR

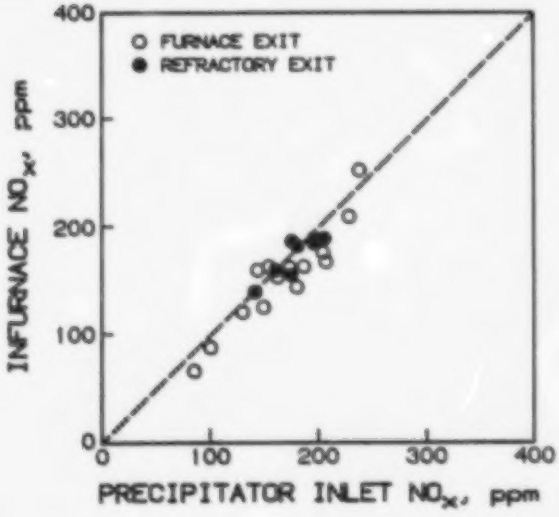

Figure 3. COMPARISON OF PRECIPITATOR INLET AND FURNACE NO_x CONCENTRATIONS

Table 2. IMPACT OF OFA ON NO_x AND COMBUSTIBLES AT THE FURNACE EXIT

OFA Used	Excess Air, %	NO_x, ppm	CO, ppm	THC, ppm
Normal	>60	125-175	20-30	0-15
None	57	88	130	30
None	23	45	1300	750

The inverse relationship between NO_x and CO is further illustrated in Figure 4. The implication here is that, if the system is to be operated for low CO evolution and, therefore, with greater assurance of lower trace polychlorinated hydrocarbons emissions, then higher NO_x must be accepted. However, the NGR technique that is being developed is believed to provide low NO_x emissions as well as improved combustible burnout. Figure 5 shows the variation in combustibles and gas temperatures along the furnace path. The gas temperatures were calculated from available in-furnace temperature traverse data combined with gas flow and heat-transfer estimates for normal furnace operation. The total residence time in the furnace is estimated to be greater than 5 seconds; the time in the intensely mixed OFA region is about 0.8 seconds; and the available time over 1800°F is approximately 2.5 seconds. From these parameters it is apparent that, with adequate mixing, excellent combustion should be achieved.

At the normal operating conditions, the gas composition data were acquired in traverses (in duplicate) about 4 feet above each of the three grates. The average of these data are presented in Table 3. The data show that the drying grate, which was supplied with minimal air, was essentially pyrolyzing the waste as evident from the very high combustible and very low O_2 concentrations. The gas phase burnout was far from complete at the measurement location above the combustion grate, which is supplied with the majority of UGA. The data also show that a small amount of combustibles remained in the gas phase above the burnout grate at a very high oxygen concentration.

Table 3. GRATE MEASUREMENT DATA (As Measured)

	Drying	Combustion	Burnout
O_2, %	0.9	10.9	20.5
NO_x, ppm	83	85	0
CO, %	10.1	5.1	20 ppm
THC, %	>1.0	>1.0	2700 ppm

The temperature and concentration data presented here have been averaged over time and furnace cross-section. The continuous point measurements, however, showed significant fluctuations with time (± 150°F) as well as variation across the furnace (160°F). Natural gas injection in the lower furnace region is expected to create a more uniform environment within the

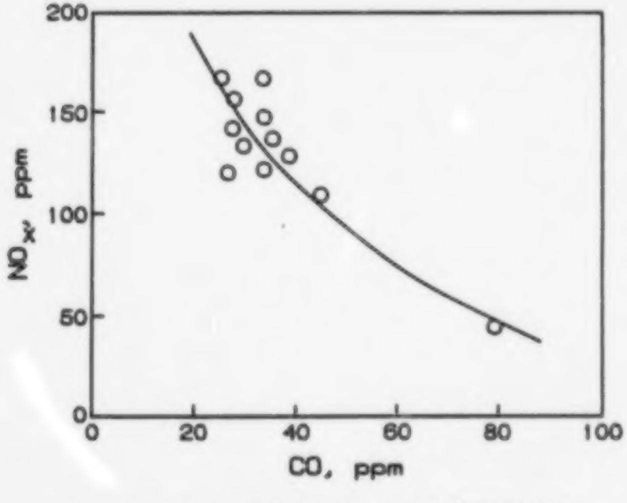

Figure 4. RELATIONSHIP BETWEEN NO_x AND CO EMISSIONS

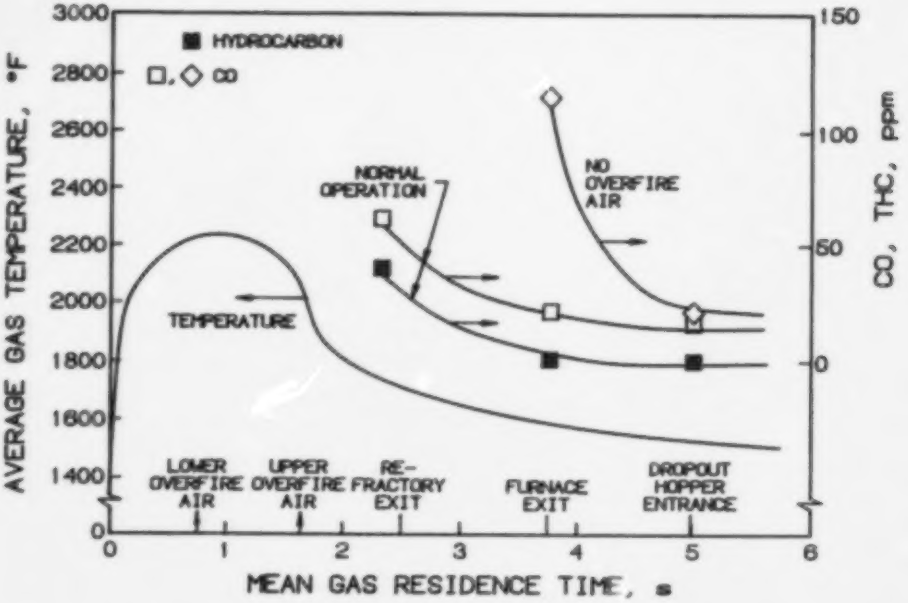

Figure 5. GAS TEMPERATURE AND COMBUSTABLES CONCENTRATIONS HISTORY IN RILEY-TAKUMA COMBUSTION SYSTEM

furnace, thereby improving combustible burnout.

The results of the baseline field testing showed that NGR, as envisioned, has good potential to reduce NO_x as well as CO and THC emissions. These data were used to design the pilot-scale experiments using simulated MSW combustion products on IGT's pilot-scale furnace, and to design and then to set the operation of the pilot-scale MSW combustor at Riley.

PILOT-SCALE TESTING

ASSESSMENT TESTS ON IGT'S PILOT-SCALE FURNACE SIMULATOR

Facility Description, Test Arrangements, and Testing

The initial pilot-scale developmental effort was carried out at the Applied Combustion Research Facility (ACRF) located at IGT's Energy Development Center (EDC). This study was undertaken to investigate the potential for achieving satisfactory NO_x reduction using NGR at the relatively lower furnace temperatures (compared to coal-fired boilers) and in time consistent with furnace residence time typical for MSW incinerators. The effectiveness of NGR for NO_x reduction was investigated on pilot-scale furnace No. 1 (overall view see Figure 6) using simulated MSW combustion products. The combustion products were simulated based on the baseline field measurements described in the previous section. The furnace has internal dimensions of 4-1/2 feet wide by 3 feet high by 14 feet long. It is equipped with removable cooling tubes along the sidewalls as well as cooling panels on the hearth to simulate a variety of load configurations. For these tests, the hearth panels were covered with insulation, and only four of the thirty cooling tubes were used along the sidewall to simulate MSW furnace load. This highly flexible furnace also has 32 removable doors along one of the sidewalls that allow insertion of diagnostic probes and injectors at almost any location in the furnace.

Figure 7 shows the schematic drawing of the test arrangement. Combustion products that would result from firing of MSW at a rate of 1.7 X 10^6 Btu/h were simulated by firing No. 2 fuel oil with the preheated combustion air in an in-line burner, adding appropriate amounts of water, oxygen, and ammonia, and firing additional fuel oil downstream to achieve the desired composition. The ammonia is used to simulate nitrogen compounds evolved on the stoker from "fuel-bound" nitrogen in the waste. The combustion products temperature was adjusted by controlling the level of combustion air preheat. The hot combustion products were introduced into the furnace through a port at one end and distributed across the furnace cross-section by a refractory distribution grid. Natural gas for reburning was mixed with cooled (~150°F) recirculated flue gases prior to injection into the furnace. The mixture was injected through holes in the top and bottom of a 4-inch-diameter stainless steel pipe injector. About 15% to 25% recirculated flue gases are anticipated to be required in the commercial unit to inject the relatively small amount of natural gas across the furnace for proper coverage and mixing with the MSW combustion products. The natural gas/flue gas mixture injector was placed

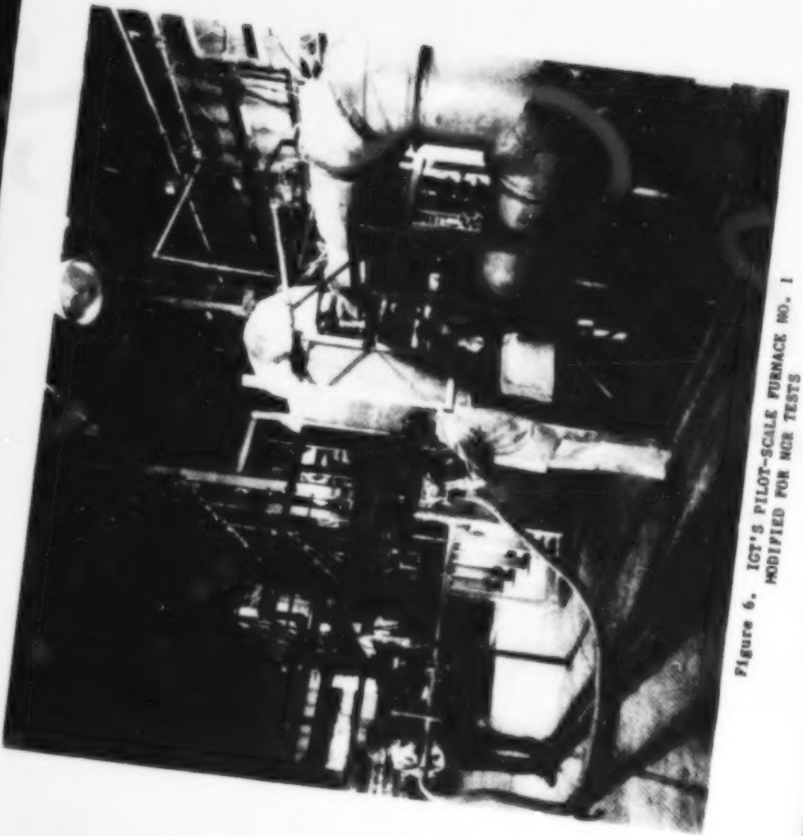

Figure 6. IGT'S PILOT-SCALE FURNACE NO. 1 MODIFIED FOR NGR TESTS

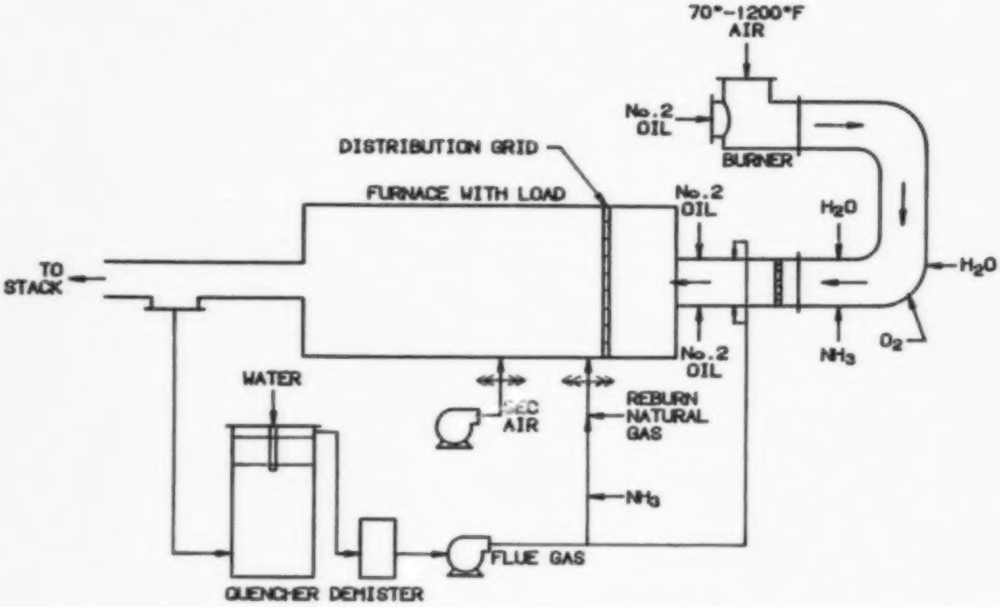
70°-1200°F
AIR
No.2
OIL
BURNER
DISTRIBUTION GRID
FURNACE WITH LOAD
No.2
OIL
H_2O
H_2O
TO
STACK
NH_3
O_2
No.2
OIL
WATER
SEC
AIR
REBURN
NATURAL
GAS
NH_3
FLUE GAS
QUENCHER
DEMISTER

Figure 7. SCHEMATIC DRAWING FOR NGR TESTS ON IGT'S PILOT-SCALE FURNACE SIMULATOR

across the furnace width at mid-height just downstream of the distribution grid. A similar injector was used for introduction of the secondary air (OFA).

The following variables were investigated under controlled conditions:

- Reburn zone residence time
- Reburn zone stoichiometry
- Reburn zone temperature
- Amount of natural gas
- NH_3 content (to simulate nitrogen compounds evolved from the waste)
- Overall stoichiometry
- Level of FGR.

The reburn zone residence time was varied in the range of 0.8 to 5.2 seconds by moving the secondary air injector along the furnace length; the reburn zone stoichiometry varied from 0.6 to 1.3 by changing the amounts of preheated combustion air and natural gas; and the reburn zone temperature was varied by adjusting the preheated combustion air temperature. The amount of natural gas, ammonia content of the reburn zone, and the level of FGR were controlled by the respective flow rates, and the overall stoichiometry by the amount of secondary air.

Table 4 compares the significant parameters in the pilot-scale simulator at typical operating conditions with those from the baseline field test unit at the design operating conditions. It was seen that, in general, the simulation was quite good. The combustion products composition and temperature, and the available residence times in the furnace simulator, were close to the values from the commercial unit.

Table 4. SIMULATION COMPARISON

	Baseline Field Test Unit, Typical	IGT Furnace Simulator, Typical
Combustion Products		
O_2, %	7.9	7.9
CO_2, %	10.2	9.8
N_2, %	71.2	70.7
H_2O, %	10.7	11.6
NO_x, ppm	130	100-350
Reburn Zone Temperature, °F	~2220	1950-2400
Reburn Residence Time, s	3.0	1.0-4.5

The primary focus of the measurements was on the temperature and composition of the combustion products in the furnace flue. The temperatures were measured using a water-cooled suction pyrometer equipped with type R thermocouples within a ceramic radiation shield. The concentrations of O_2, CO, THC, CO_2, and NO_x were measured using a water-cooled sampling probe connected to continuous analyzers. Gas temperatures and compositions were also measured in the reburn zone; however, these are not presented here.

Test Results

The NO_x emissions during the tests showed significant variations with the operating conditions while the CO and the THC emissions were generally below 20 ppm and 5 ppm, respectively. The very low level of combustibles resulted from the excellent mixing characteristics of the injectors. A high level of mixing was provided intentionally to avoid the impact of mixing on analysis of the data because the mixing characteristics that exist in the commercial mass burn units could not be duplicated on the pilot-scale simulator.

In one test series, however, poor fuel oil atomization resulted in CO levels in the range of 30 to 40 ppm in the flue gases. Injection of 15% natural gas reduced this CO to below 20 ppm indicating the potential for CO reduction with NGR.

The measurement of prime interest was the effect of NGR on NO_x emissions in the flue gases. These NO_x data are presented in Figures 8 through 12. The NH_3 concentrations shown in these figures represent the NH_3 injected directly into the reburn zone. FGR is expressed as a percentage of flue gases from the furnace exhaust. And the amount of natural gas is expressed as a percentage of MSW heat input. The grate stoichiometric ratio (GSR) is the stoichiometric ratio of the combustion products entering the reburn zone, whereas the reburn stoichiometric ratio (RSR) is the stoichiometric ratio after addition of the natural gas/flue gas mixture into these gases. The reburn residence time is the time between the natural gas/flue gas mixture injection and the secondary air injection. Figure 8 shows that with a GSR of 1.0 entering the reburn zone, NO_x decreased with increasing the natural gas level injection as well as reburn residence time. Compared to levels of 200 to 225 ppm with no natural gas addition, NO_x decreased to 125 to 75 ppm with 15% natural gas. NO_x decreased with increasing the reburn residence time, however, residence time longer than 2.5 seconds had little effect on further NO_x reduction.

The significance of reburn residence time on NO_x reduction is further illustrated in Figure 9. The NO_x remaining is expressed as a percentage of the corresponding NO_x concentration prior to natural gas injection. For example, without natural gas, the NO_x emission was 100 ppm. With 15% natural gas, NO_x would drop to about 60 ppm if a reburn zone residence time of 1 second is used, to about 40 ppm at 2.5 seconds, and to about 35 ppm at 4.5 seconds. Figure 10 shows the impact of the GSR on the effectiveness of NGR. The figure shows that injection of 15% natural gas was very effective in reducing NO_x emissions for GSR of between 1 and 1.2 — a condition at which most MSW systems are designed to operate. Although the actual NO_x levels

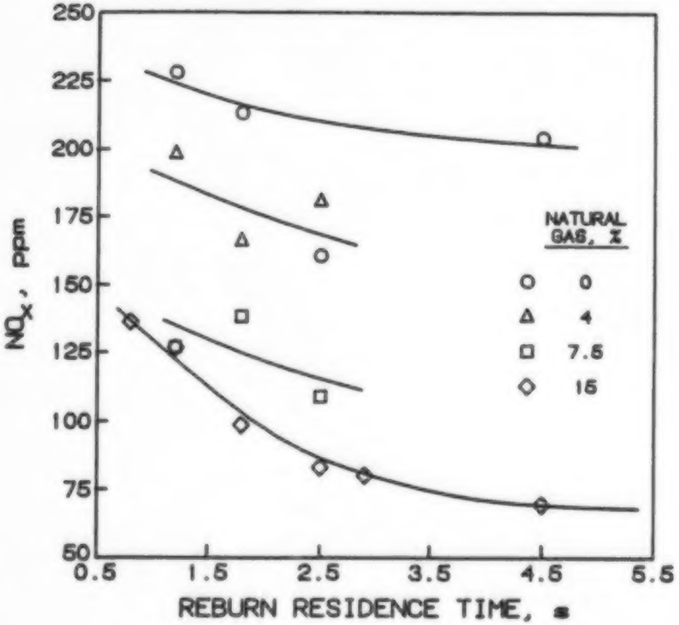

Figure 8. EFFECT OF NATURAL GAS INJECTION AND REBURN ZONE RESIDENCE TIME ON NO$_x$ EMISSIONS (NH_3: 1000 ppm, FGR: 17%, GSR: 1.0)

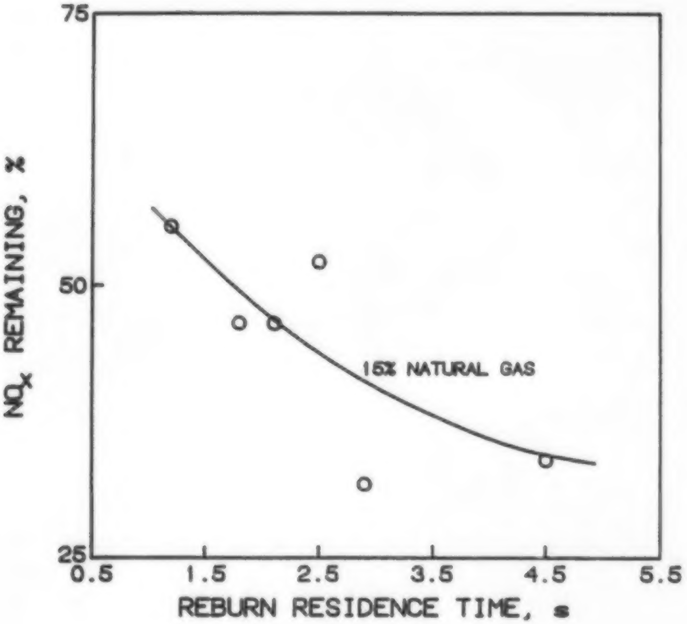

Figure 9. EFFECT OF REBURN RESIDENCE TIME ON NO$_x$ REDUCTION WITH NATURAL GAS INJECTION (NH$_3$: 1000 ppm, FGR: 17%, GSR: 1.0)

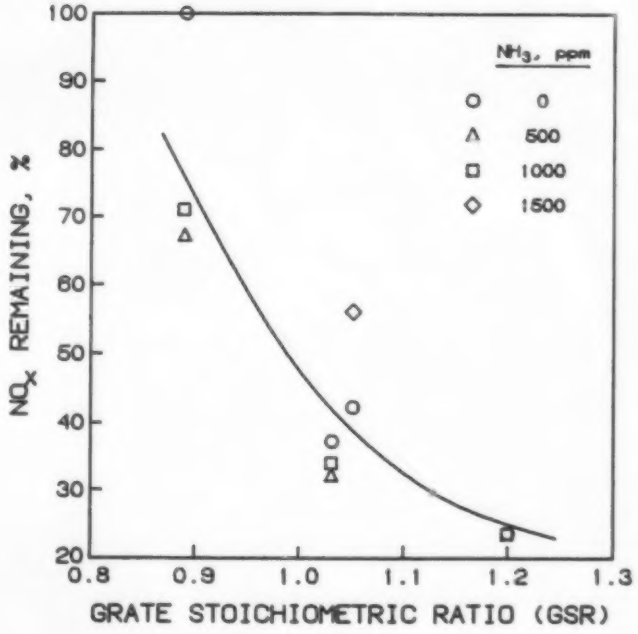

Figure 10. EFFECT OF GSR ON NO$_x$ REDUCTION
(Residence Time: 4 to 5.2 s, Reburn
Temperature: 2050°F, FGR: 17%,
Natural Gas: 15%)

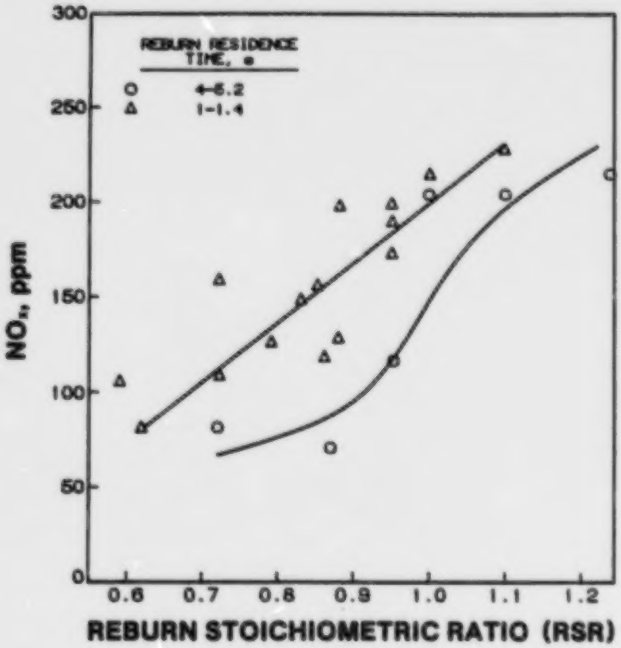

Figure 11. EFFECT OF RSR ON NO_x EMISSIONS (NH_3: 1000 ppm, FGR: 17%)

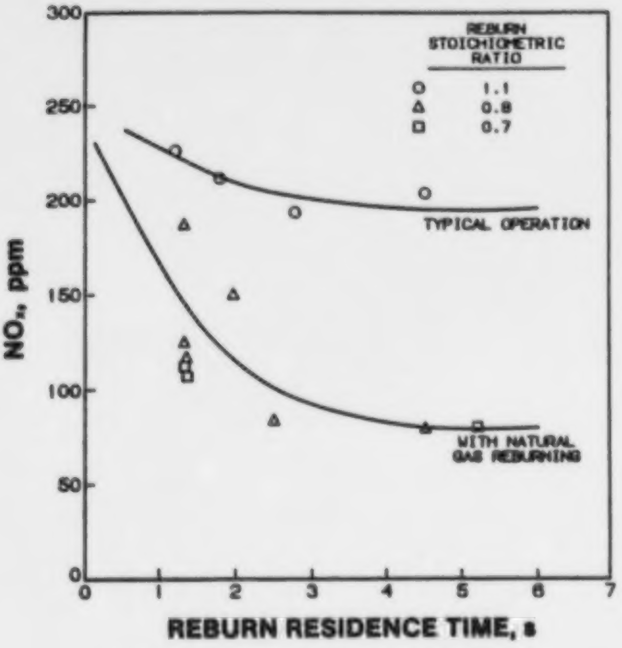

Figure 12. REBURN RESIDENCE TIME REQUIREMENT FOR EFFECTING NO_x REDUCTION (NH_3: 1000 ppm, FGR: 17%)

increased significantly with increasing the amount of NH_3 added to the reburn zone (especially at low NH_3 injection levels), the percent reduction in NO_x was relatively insensitive to the NH_3 concentrations. Without NH_3 addition to the combustion products, the only NO_x formed was during the combustion of No. 2 fuel oil. Because this NO_x was formed prior to the natural gas injection, any decrease in NO_x represents actual destruction of NO_x to molecular nitrogen. With NH_3 addition directly into the reburn zone, the decrease in flue gas NO_x concentration represents not only destruction of NO_x that had already formed, but also the decomposition of NH_3, part of which would have otherwise oxidized to NO_x upon injection of the secondary air.

The effect of RSR on NO_x emissions is illustrated in Figure 11. These data represent variations in the RSR caused by changing the amount of natural gas injection as well as GSR. A RSR of between 1.1 and 1.2 gives typical operation without natural gas injection. The NO_x levels at these conditions were about 220 ppm. Injection of 15% natural gas would reduce RSR to about 0.9. The corresponding NO_x value was about 70 ppm, or 70% lower, with about 5 seconds spent in the reburn zone. With reburn residence time between 4 and 5.2 seconds, reduction of RSR to below 0.9 had no effect on further NO_x reduction. With smaller residence times in the range of 1 to 1.4 seconds, the NO_x reduction at RSR of about 0.9 was not as significant. In this case, NO_x was reduced to about 150 ppm — a 35% reduction.

Figure 12 further illustrates the effectiveness of reburn residence time in reducing the NO_x emissions by NGR. With an injection of natural gas sufficient to create a RSR of 0.9, the NO_x emissions decreased rapidly with increasing the residence time. At a residence time of 2.5 seconds, the NO_x emissions were less than 40% of the original value. Further increases in the residence time in the reburn zone had little effect on NO_x emissions.

Another parameter that was believed to be of significant importance in determining the effects of NGR was the temperature in the reburn zone. This temperature was varied during the tests from 1950° to 2400°F. Some of the variations were obtained by changing the air preheat temperature; however, most of them were caused by variations in the other operating conditions such as the GSR, the amount of NGR, and the amount of FGR. Even with the relatively larger variations in the temperature, its effect on NO_x emissions were not obvious. The temperature effects were probably small and were masked by the impact of other, more significant parameters.

The results of the pilot-scale testing at IGT using simulated MSW combustion products can be summarized as follows:

- Up to 70% reduction in NO_x was measured when natural gas was injected into simulated products of MSW combustion to create reducing conditions. A RSR of 0.9, that was created by injection of 15% natural gas, and which should be feasible for the commercial systems, was sufficient. Further reductions in the RSR were not beneficial.

- A reburn zone residence time of about 2.5 seconds at temperatures above 1900°F was required. The typical commercial unit has a residence time of more than 3 seconds available above 1900°F. Residence times larger than 2.5 seconds did not provide further significant NO_x reductions.

- Some of the test data show the potential for simultaneous reductions in NO_x and CO emissions. This is significant because of the implied relationship between CO and trace polychlorinated hydrocarbons emissions.

- The reburn zone temperature in the range tested does not appear to be a critical parameter in determining the effectiveness of NGR.

INITIAL TESTS ON RILEY'S PILOT-SCALE MSW COMBUSTOR

Facility Description, Test Arrangements, and Testing

Following the encouraging results from the furnace simulator, NGR strategies were further evaluated on a 3 X 10^6 Btu/h pilot-scale combustion facility at Riley's Research Center (6). Originally designed for testing coal combustion, the facility was modified for firing processed MSW by installing a scaled, stepped Takuma stoker. The 17-foot-high by 12-foot-deep by 3-foot-wide combustor/furnace, shown schematically in Figure 13, is designed to fire up to 6 ton/day of processed MSW (a reduced size MSW with most glass and metals having been removed). The processed MSW, stored in a 10-ton trailer, was loaded into the feed chute hopper by a 45-foot-long drag chain conveyor. The waste was then fed into the combustion chamber through the feed chute by the hydraulic feed pushers. The pilot-scale combustor is equipped with an ash discharge hopper, independent UGA and OFA supplies, FGR system, and multiple levels of natural gas and air injectors to facilitate testing. The water-cooled furnace walls are lined with insulating refractory to simulate the proper thermal environment.

The furnace chamber has multiple ports in the sidewalls for gas temperature and gas composition measurements. A computerized data acquisition system is used for direct on-line data analysis. In-furnace gas temperature measurements are made with suction pyrometer probes, while the concentrations of O_2, CO_2, NO_x, CO, THC, and SO_x are measured using gas sample probes and continuous analyzers.

The initial testing was directed at characterizing the operation to ensure proper simulation of the full-scale unit operation. The impact of firing rate, UGA and OFA distribution, and excess air on the combustor performance was studied to achieve the desired conditions. Table 5 compares the characteristics of pilot-scale unit operation with the data from the baseline field test. The agreement is quite good, the only major difference being in the available gas residence times. The commercial unit had twice as long available residence times; this should help the NGR process.

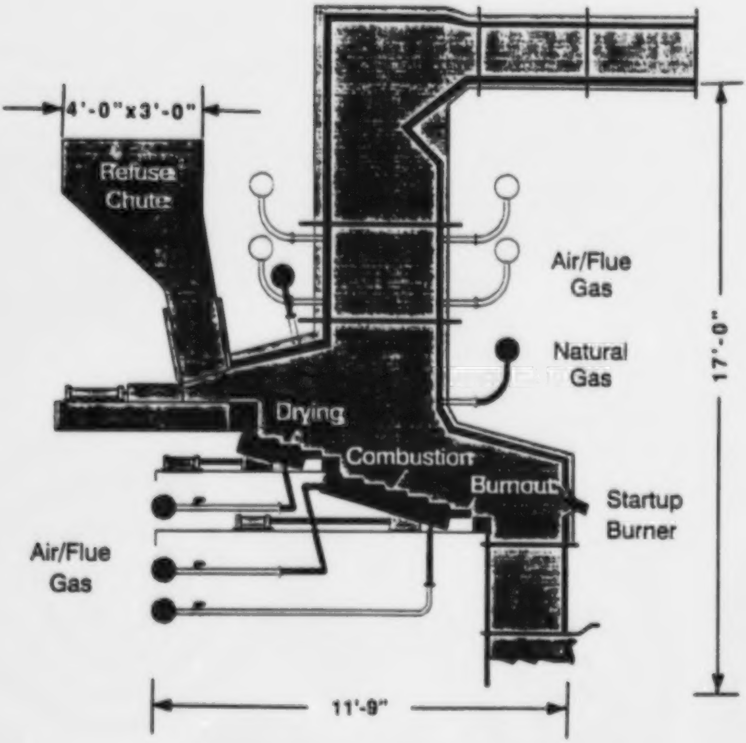
4'-0"x3'-0"
Refuse Chute
Air/Flue Gas
Natural Gas
17'-0"
Drying
Combustion
Burnout
Startup Burner
Air/Flue Gas
11'-9"

Figure 13. RILEY PILOT-SCALE MSW COMBUSTION FACILITY

Table 5. COMPARISON OF PILOT-SCALE TEST DATA WITH BASELINE FIELD TEST DATA

	Field Test Unit	Riley Pilot Unit
Test No.	21	7
MSW Heating Value, Btu/lb	6037	5447
Load, 10^6 Btu/h	37.5	2.36
Excess Air, %	73	70
OFA Flow, %	34	38
OFA Configuration	Standard	Standard
Combustion Products		
O_2, %	9.3	8.7
CO_2, %	10.1	10.9
NO_x, ppm	134	142
CO, ppm	29	27
THC, ppm	0	0
Furnace Exit Temperature, °F	1620	1570
Reburn Zone Temperature, °F[a]	2220	2181
Residence time to Furnace Exit, s	~3.8	~2.0

[a] Calculated from measured Furnace Exit Temperature

Once the desired system performance was achieved, testing was directed at studying NGR strategies. Initial tests focused on studying the impacts of the amounts and injection locations of natural gas, FGR, and OFA. Effects of furnace stoichiometry were also investigated.

Test Results

As the emissions of NO_x, CO, and THC were of primary concern, only those results are presented here. The results of initial tests with reburn residence time of less than 1 second were encouraging. With injection of 15% natural gas, it was possible to reduce NO_x levels from a range of 120 to 165 ppm to about 70 to 80 ppm, or nearly 50%. This effect is further illustrated in Figure 14. The data represent a range of firing rates, overall excess air levels, grate stoichiometries, and injector locations. In all cases, NO_x emissions were less with NGR. The overall excess air was generally lower, about 40%, with NGR, while emission levels of CO and THC were either comparable or lower than the levels without natural gas injection.

Figure 15 shows the significance of reburn zone residence time in reducing NO_x emissions. An additional 0.2 second residence time in the reburn zone decreased the NO_x by 10%. As discussed earlier, the commercial units can have residence times of over 2 seconds available for NGR, and tests on the furnace simulator had also shown significantly greater reduction in NO_x with residence times of about 2.5 seconds. The next series of tests will, therefore, investigate the effectiveness of NGR at larger residence times. Up to 1.5 seconds are available on the pilot-scale unit.

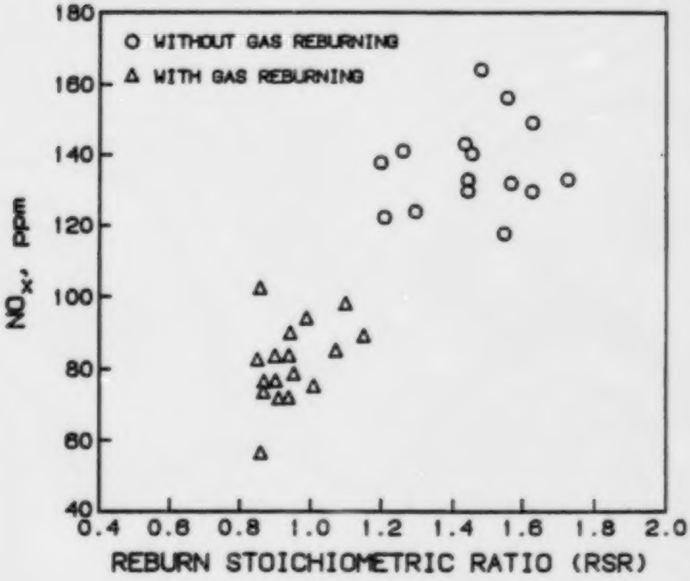

Figure 14. EFFECT OF RSR ON NO_x EMISSIONS

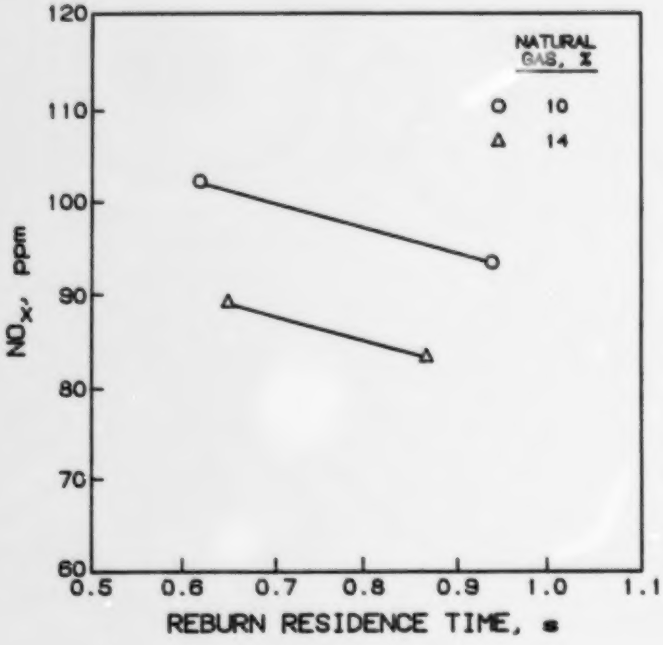

Figure 15. EFFECT OF NATURAL GAS INJECTION AND REBURN ZONE RESIDENCE TIME ON NO$_x$ EMISSIONS

The major findings from the initial NGR tests on the pilot-scale MSW combustor are as follows:

- A RSR of approximately 0.9 is effective in reducing NO_x while residence times longer than 1 second are required.
- Effective burnout of CO and THC emissions can be achieved with NGR.
- It may be possible to reduce the overall excess air from the existing 70% to 30% without adversely affecting emissions.
- The furnace exhaust gas temperatures did not increase appreciably upon injection of natural gas.

FUTURE EFFORT

Testing on Riley's Pilot-Scale MSW Combustion Facility will continue to further optimize the NGR system. Test variables will focus primarily on increasing reburn zone residence time and improving mixing, especially near the stoker, in an effort to obtain >50% NO_x reduction while maintaining excellent CO and THC burnout. Modifications will be made to introduce OFA as high in the furnace as possible in order to maximize residence time. This will provide the capability of testing residence times >1.5 seconds. The best injection strategy determined during the initial testing will continue to be studied, particularly toward reduction of the amount of both natural gas and recirculated flue gases.

Following the determination of the final gas reburning configuration, more detailed in-furnace testing will be performed to better characterize the gas composition and temperature history throughout the furnace both with and without gas reburning. If possible, dioxin measurements will be collected.

Assuming success with the final pilot-scale testing, plans for a field test of this technology in a commercial operating facility will be finalized. A detailed economic analysis of the gas reburning technology will first be conducted to determine the economic advantage of utilizing natural gas for emissions control. Boiler performance analysis will also be performed to quantify the impact of a gas reburning system on overall boiler performance. If this final analysis is encouraging, a field test program of the gas reburning technology will be undertaken.

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